SUMMARY OF PEER REVIEW COMMENTS
ON THE 2015 GREEN TURTLE STATUS REVIEW

In January 2014, NOAA Fisheries and the U.S. Fish and Wildlife Service (together the Services) invited 15 sea turtle experts to peer review the Green Turtle (*Chelonia mydas*) Status Review Under the U.S. Endangered Species Act. In this document, the Services have compiled all peer review comments (made by the following peer reviewers but not identified to any one peer reviewer) and our response to their comments. The final Status Review benefited from the peer review process, and we would like to thank the following reviewers for their thoughtful contributions:

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EXECUTIVE SUMMARY

The green turtle (Chelonia mydas) was listed under the ESA on July 28, 1978. Breeding populations of the green turtle in Florida and along the Pacific Coast of Mexico were listed as endangered; all other populations were listed as threatened. In 2007, the National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (FWS); together the Services) completed a 5-year review for the green turtle. A 5-year review is an assessment of a listed species to determine whether its status has changed since the time of its listing such that it should be delisted or classified differently than its current status. The Services concluded that new information available since the completion of the previous reviews indicated a possible separation of populations by ocean basins but that a more in-depth analysis was needed to determine the application of the distinct population segment (DPS) policy. Based on the new information and the need for further analysis under the DPS policy, the Services recommended that no change in listing status was warranted in 2007. However, they committed to fully assemble and analyze all relevant information in accordance with the DPS policy.

On February 16, 2012, the Services received a petition from the Association of Hawaiian Civic Clubs to identify the Hawaiian green turtle population as a DPS and delist the DPS under the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. § 1531 et seq.). On August 1, 2012, NMFS (with FWS concurrence) determined that the petition presented substantial information indicating that the petitioned action may be warranted and initiated a status review to determine whether the petitioned action is warranted. The Services decided to review the Hawaiian population in the context of examining green turtles globally with regard to application of the DPS policy and in light of significant new information since the listing of the species in 1978. This is consistent with the recommendation in the 2007 review.

The Services convened a status review team (SRT) in November 2012 to review the best available scientific information, determine whether DPSs exist, and assess the extinction risk for any identified DPS. In accordance with the DPS policy, a population may be defined as a DPS if it is both discrete and significant relative to its taxon.

With regard to discreteness, the SRT evaluated genetic evidence, tagging (flipper and passive integrated transponder (PIT) tags) and satellite telemetry data, demographics information, oceanographic features, and geographic barriers. It determined that there are 11 discrete population segments for green turtles globally. These discrete population segments are markedly separated from each other as a consequence of ecological, behavioral, and oceanographic factors, and based on genetic and morphological evidence.

The SRT then considered whether each of the 11 identified discrete population segments is significant relative to its taxon. The SRT determined that each of the 11 discrete population segments were biologically and ecologically significant. They each represent a large portion of the species range, whose loss would result in a significant gap in distribution of the species. Each discrete population segment is genetically unique; the loss of any one discrete population segment would represent a significant loss of genetic diversity. Further, some DPSs represent unique ecological settings influenced by local ecological and physical factors, some exhibit unique morphological or other demographic characteristics, and others have unique movement
patterns. Therefore, the SRT concluded that the 11 identified population segments are both
discrete from other conspecific population segments and significant to the species, *Chelonia mydas*. Although DPS is a legal term and the SRT recognizes that these population segments are
not technically DPSs until or unless they are designated as such in a rulemaking process, for lack
of a better term, we refer to these units as DPSs throughout the report.

The SRT identified the following 11 green turtle DPSs distributed globally:

(1) North Atlantic DPS
(2) Mediterranean DPS
(3) South Atlantic DPS
(4) Southwest Indian DPS
(5) North Indian DPS
(6) East Indian - West Pacific DPS
(7) Central West Pacific DPS
(8) Southwest Pacific DPS
(9) Central South Pacific DPS
(10) Central North Pacific DPS
(11) East Pacific DPS

After the 11 DPSs were identified, the SRT assessed the extinction risk for each DPS. Six
critical assessment elements were considered and quantified in this assessment: (1) abundance;
(2) population growth rate or productivity; (3) spatial structure; (4) diversity / resilience; (5)
threats (as represented by the five factors in section 4(a)(1) of the ESA); and (6) conservation
efforts. Each SRT voting member ranked the importance of each of the population elements
(first four above) by assigning them a value from 1 to 5, with 1 representing a very low risk.
They ranked the influence of the five factors (threats) on the status of the DPS by assigning a
value of 0 (neutral) to –2, and ranked the influence of conservation efforts on the status of the
DPS by assigning a value of 0 to 2. The SRT noted that none of these elements is entirely
independent, and did not attempt to use the values applied to each element by each SRT member
to arrive at extinction risk.

In the next step, each SRT voting member gave their expert opinion on the likelihood that each
DPS would reach critical risk thresholds within 100 years by spreading 100 points across several
risk categories for each DPS. Finally, the SRT reviewed information on threats and extinction
risk to portions of the ranges for each DPS that, at present, have substantially higher risk than
other parts of the DPS and evaluated if these are significant. **Only two DPSs were found to
potentially have significant portions of their ranges (SPRs), the Central North Pacific DPS, and
the East Indian-West Pacific DPS.** The SRT conducted two votes for the risk of extinction to
these DPSs: One for the entire DPS, and one for the DPS that would remain if the SPR is lost.

A summary of the SRT’s discussions of the critical assessment elements, overall risk of
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LIST OF ACRONYMS AND ABBREVIATIONS

The following are standard abbreviations for acronyms and terms found throughout this
document:

ANET  Andaman and Nicobar Island Environmental Team
ASEAN  Association of South East Asian Nations
BCS  Baja California Sur
CBD  Convention on Biological Diversity
CBRA  Coastal Barrier Resources Act
CCL  Curved Carapace Length
CL  Carapace Length
CI  Confidence Interval
cm  centimeter
<table>
<thead>
<tr>
<th>Acronym</th>
<th>Full Form</th>
</tr>
</thead>
<tbody>
<tr>
<td>CMAR</td>
<td>ETP Marine Corridor Initiative</td>
</tr>
<tr>
<td>CMS</td>
<td>Convention on the Conservation of Migratory Species of Wild Animals</td>
</tr>
<tr>
<td>COFI</td>
<td>FAO Committee on Fisheries</td>
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<tr>
<td>DNA</td>
<td>Deoxyribonucleic Acid</td>
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<tr>
<td>DPS</td>
<td>Distinct Population Segment</td>
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<tr>
<td>EEZ</td>
<td>Exclusive Economic Zone</td>
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<tr>
<td>ENSO</td>
<td>El Niño Southern Oscillation</td>
</tr>
<tr>
<td>ESA</td>
<td>Endangered Species Act of 1973, as amended</td>
</tr>
<tr>
<td>ESU</td>
<td>Evolutionary Significant Unit</td>
</tr>
<tr>
<td>ETP</td>
<td>Eastern Tropical Pacific</td>
</tr>
<tr>
<td>FAO</td>
<td>Food and Agriculture Organization of the United Nations</td>
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<tr>
<td>FP</td>
<td>Fibropapillomatosis</td>
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<tr>
<td>FFS</td>
<td>French Frigate Shoals</td>
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<tr>
<td>FWC</td>
<td>Florida Fish and Wildlife Conservation Commission</td>
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<tr>
<td>FWS</td>
<td>U.S. Fish and Wildlife Service</td>
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<td>FST</td>
<td>Genetics Fixation Index</td>
</tr>
<tr>
<td>GBR</td>
<td>Great Barrier Reef</td>
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<tr>
<td>GIWA</td>
<td>Global International Waters Assessment</td>
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<tr>
<td>km</td>
<td>kilometer</td>
</tr>
<tr>
<td>IAC</td>
<td>Inter-American Convention for the Protection and Conservation of Marine Turtles</td>
</tr>
<tr>
<td>IATTC</td>
<td>InterAmerican Tropical Tuna Commission</td>
</tr>
<tr>
<td>ICAT</td>
<td>International Commission for the Conservation of Atlantic Tunas</td>
</tr>
<tr>
<td>IOSEA</td>
<td>Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding</td>
</tr>
<tr>
<td>IOTC</td>
<td>Indian Ocean Tuna Commission</td>
</tr>
<tr>
<td>IPCC</td>
<td>Intergovernmental Panel on Climate Change</td>
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<tr>
<td>IRG</td>
<td>In-water Research Group</td>
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<tr>
<td>IUCN</td>
<td>International Union for the Conservation of Nature</td>
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<tr>
<td>l</td>
<td>liter</td>
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<tr>
<td>LME</td>
<td>Large Marine Ecosystem</td>
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<tr>
<td>m</td>
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<tr>
<td>MARPOL</td>
<td>International Convention for the Prevention of Pollution from Ships</td>
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<tr>
<td>ml</td>
<td>milliliter</td>
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<tr>
<td>MNS</td>
<td>Mean Nesting Size</td>
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<tr>
<td>MOU</td>
<td>Memorandum of Understanding</td>
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<tr>
<td>MPA</td>
<td>Marine Protected Area</td>
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<tr>
<td>MSA</td>
<td>U.S. Magnuson-Stevens Fishery Conservation and Management Act</td>
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<tr>
<td>mya</td>
<td>million years ago</td>
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<tr>
<td>MHI</td>
<td>Main Hawaiian Islands</td>
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<tr>
<td>mtDNA</td>
<td>mitochondrial Deoxyribonucleic Acid</td>
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<td>MX</td>
<td>Mexico</td>
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<tr>
<td>nDNA</td>
<td>nuclear Deoxyribonucleic Acid</td>
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<td>nGBR</td>
<td>northern Great Barrier Reef</td>
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<tr>
<td>NGO</td>
<td>Non-governmental organization</td>
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<td>NMFS</td>
<td>National Marine Fisheries Service</td>
</tr>
<tr>
<td>Acronym</td>
<td>Definition</td>
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<tr>
<td>NPF</td>
<td>Northern Australian Prawn Fishery</td>
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<tr>
<td>NRC</td>
<td>National Research Council</td>
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<td>NWHI</td>
<td>Northwest Hawaiian Islands</td>
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<tr>
<td>PIT</td>
<td>Passive Integrated Transponder</td>
</tr>
<tr>
<td>PVA</td>
<td>Population Viability Analysis</td>
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<tr>
<td>QET</td>
<td>Quasi-extinction Threshold</td>
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<td>RI</td>
<td>Remigration Interval</td>
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<tr>
<td>RFMO</td>
<td>Regional Fishery Management Organizations</td>
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<tr>
<td>SCL</td>
<td>Straight Carapace Length</td>
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<td>SD</td>
<td>Standard Deviation</td>
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<td>SEAFO</td>
<td>South-East Atlantic Fisheries Organization</td>
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<tr>
<td>sGBR</td>
<td>southern Great Barrier Reef</td>
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<tr>
<td>SPAW</td>
<td>Protocol Concerning Specially Protected Areas and Wildlife</td>
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<tr>
<td>SPR</td>
<td>Significant Portion of its Range</td>
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<tr>
<td>SPREP</td>
<td>South Pacific Regional Environment Programme</td>
</tr>
<tr>
<td>SWIO</td>
<td>Southwest Indian Ocean</td>
</tr>
<tr>
<td>SQE</td>
<td>Susceptibility to Quasi-extinction</td>
</tr>
<tr>
<td>SRT</td>
<td>Status Review Team</td>
</tr>
<tr>
<td>TED</td>
<td>Turtle Excluder Device</td>
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<tr>
<td>TIHPA</td>
<td>Turtle Island Heritage Protection Area</td>
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<tr>
<td>UK</td>
<td>United Kingdom</td>
</tr>
<tr>
<td>UNEP</td>
<td>United Nations Environment Programme</td>
</tr>
<tr>
<td>UNESCO</td>
<td>United Nations Educational, Scientific, and Cultural Organization</td>
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<td>yr(s)</td>
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1. INTRODUCTION AND BACKGROUND

1.1. ESA Overview

1.1.1. Purpose

The purpose of the Endangered Species Act of 1973, as amended (ESA; 16 U.S.C. § 1531 et seq.) is to provide a means to conserve ecosystems upon which endangered and threatened species depend, to provide a program for the conservation of endangered and threatened species, and to take appropriate steps to recover endangered and threatened species. The National Marine Fisheries Service (NMFS) and U.S. Fish and Wildlife Service (FWS; together, the Services) share responsibility for administering the ESA. The Services are responsible for determining whether species, subspecies, or distinct population segments of vertebrate species are threatened or endangered under the ESA. FWS typically has the lead for terrestrial and freshwater species, and NMFS typically has the lead for marine, estuarine, and anadromous species. However, the Services share jurisdiction over sea turtles under the ESA; NMFS is responsible for sea turtles in their marine environment and FWS is responsible for sea turtles in their terrestrial environment. The Services worked together on this document through participation on a Status Review Team, as discussed in Section 1.3 below.

1.1.2. Definitions

The following are definitions as defined in the ESA:

Species - includes any subspecies of fish or wildlife or plants and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature.

Endangered Species - any species which is in danger of extinction throughout all or a significant portion of its range.

Threatened Species - any species which is likely to become an endangered species within the foreseeable future throughout all or a significant portion of its range.

1.1.3. Listing

Section 4 of the ESA specifies a process for determining whether a species should be listed as threatened or endangered, changed in status from endangered to threatened or vice versa, or removed from the list. The determination is based solely on the best available scientific and commercial data available after reviewing the status of the species and taking into account conservation efforts. The Services must determine whether any species is an endangered species or a threatened species because of any of the following factors (Section 4(a)(1)(A)-(E)):

A. the present or threatened, destruction, modification, or curtailment of its habitat or range;
B. overutilization for commercial, recreational, scientific, or educational purposes;
C. disease or predation;  
D. the inadequacy of existing regulatory mechanisms; and  
E. other natural or manmade factors affecting its continued existence.

The Services can begin the review for listing determinations, or any interested person may petition for a listing determination under section 553(e) of U.S.C. title 5.

1.1.4. Distinct Population Segment

The ESA, as originally passed, defined species to include, "...any subspecies of fish or wildlife or plants and any other group of fish or wildlife of the same species or smaller taxa in common spatial arrangement that interbreed when mature." In 1978 amendments, the Act was changed to define a species as including "...any subspecies of fish or wildlife or plants, and any distinct population segment of any species of vertebrate fish or wildlife which interbreeds when mature." Thus, the term "distinct population segment," or DPS, was coined with the 1978 amendments.

In 1996, the Services published the Policy Regarding the Recognition of Distinct Vertebrate Population Segments Under the ESA (61 FR 4722, February 7, 1996). The policy defines a population to be a DPS if it is both discrete and significant relative to its taxon. A population may be considered discrete if it satisfies either one of the following conditions:

- 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.
- It is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of section 4(a)(1)(D) of the ESA.

If a population segment is considered discrete, NMFS and/or FWS must then consider whether the discrete segment is significant relative to its taxon. Criteria that can be used to determine whether the discrete population segment is significant include, but are not limited to, the following:

- Persistence of the discrete population segment in an ecological setting unusual or unique for the taxon,
- Evidence that loss of the discrete population segment would result in a significant gap in the range of the taxon,
- 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
- Evidence that the discrete population segment differs markedly from other populations of the species in its genetic characteristics.
1.2. History of Green Turtle Listing, Status Reviews, and Petitions

1.2.1. ESA Listing

The green turtle (*Chelonia mydas*) was listed under the ESA on July 28, 1978 (43 FR 32800). Breeding populations of the green turtle in Florida and along the Pacific Coast of Mexico were listed as endangered; all other populations were listed as threatened. The major factors contributing to its status included human encroachment and associated activities on nesting beaches; commercial harvest of eggs, subadults, and adults; predation; lack of comprehensive and consistent protective regulations; and incidental take in fisheries. Marine critical habitat for the green turtle was designated on September 2, 1998 (63 FR 46693) for the waters surrounding Culebra Island, Commonwealth of Puerto Rico (Puerto Rico), and its outlying keys.

1.2.2. ESA 5-year Reviews

Under the ESA, FWS and NMFS are required to conduct a review of listed species under their jurisdiction at least once every 5 years. A 5-year review is an assessment of a listed species to determine whether its status has changed since the time of its listing such that it should be delisted or its classification changed. The purpose of a 5-year review is to ensure that a listed species has the appropriate level of protection under the ESA.

FWS conducted reviews of the green turtle in 1983 (48 FR 55100, December 8, 1983) and in 1991 (56 FR 56882, November 6, 1991). In these reviews, the status of many species was simultaneously evaluated with a relatively cursory assessment of the five factors or threats as they pertain to the individual species. The notices stated that FWS was seeking any new or additional information reflecting the necessity of a change in the status of the species under review. The notices indicated that if significant data were available warranting a change in a species’ classification, the Service would propose a rule to modify the species’ status. No change in the green turtle’s listing classification was recommended from these reviews.

NMFS conducted its first review for the green turtle in 1985 (Mager, 1985). Data on population trends were limited and were based largely on the number of nests and nesting females. Of 52 nesting populations examined throughout the Atlantic, Pacific, and Indian Oceans, 33 were thought to be declining, 18 were unknown, and only one—the southeast U.S. Atlantic—was thought to be increasing. Although commercial harvest of eggs had decreased and the U.S. had implemented protective regulations, many threats continued both domestically and abroad. NMFS concluded that the listing as endangered in Florida and on the Pacific coast of Mexico and threatened in the rest of its range was still appropriate and should be retained.

In 1995, the Services conducted a joint review on the East Pacific green turtle only (Plotkin, 1995). The conclusion was to retain the listing of that population as endangered throughout its range.

The last review was conducted jointly and completed in 2007 (NMFS and USFWS, 2007). Many technological advances and a diversity of research had occurred since the last reviews. Molecular markers (i.e., mitochondrial DNA (mtDNA) and microsatellites) helped define the
genetic structuring within and among ocean basins, both at nesting beaches and at foraging
grounds. New information existed on demographic parameters such as age at first reproduction
and survival rates and the biology of green turtles, especially away from the nesting beach.
These data indicated a possible separation of populations by ocean basins; however, a more in-
depth analysis was needed to determine the application of the DPS policy. Based on the new
information and the need for further analysis under the DPS policy, the Services recommended
that no change in listing status was warranted. However, they committed to fully assemble and
analyze all relevant information in accordance with the DPS policy.

1.2.3. Recovery plans

The following are the recovery plans that have been developed for green turtles:

**Name of plan:** Recovery Plan for the U.S. Population of Atlantic Green Turtle (*Chelonia mydas*)
**Date issued:** October 29, 1991

**Name of plan:** Recovery Plan for the U.S. Pacific Populations of the Green Turtle (*Chelonia mydas*)
**Date issued:** January 12, 1998

**Name of plan:** Recovery Plan for the U.S. Pacific Populations of the East Pacific Green Turtle (*Chelonia mydas*)
**Date issued:** January 12, 1998
**Date of previous plan:** Original plan date - September 19, 1984

1.2.4. Petition

On February 16, 2012, the Services received a petition from the Association of Hawaiian Civic
Clubs to identify the Hawaiian green turtle population as a DPS and delist the DPS under the
ESA. On August 1, 2012, NMFS (with FWS concurrence) determined that the petition presented
substantial information indicating that the petitioned action may be warranted (77 FR 45571,
August 1, 2012), and initiated a status review to determine whether the petitioned action is
warranted. The Services decided to review the Hawaiian population in the context of examining
green turtles globally with regard to application of the DPS policy and in light of significant new
information since the listing of the species in 1978. This is consistent with the recommendation
in the 2007 review (see Section 1.2.2).

1.3. Status Review Team

The Services appointed a Status Review Team (SRT) in September 2012 and convened the SRT
for the first time in November 2012 to review the best available scientific information, determine
whether DPSs exist, and assess the extinction risk for any identified DPS. The SRT was an
advisory group to the Services and consisted of members from both agencies. SRT members
were affiliated with NMFS Science Centers, NMFS and FWS Regional Offices, and NMFS and
FWS Headquarters Offices, and provided expert knowledge to the SRT on a diverse
range of expertise, including green turtle population structure, biology, demography, ecology, and management challenges, as well as expertise on risk analysis and ESA policy. The SRT Chair, chosen by the team, was Jeffrey Seminoff (NMFS, Southwest Fisheries Science Center). This report is the outcome of the SRT findings and best expert opinion.
2. SPECIES OVERVIEW

2.1. Taxonomy

The scientific classification of the green turtle is:

Kingdom: Animalia
Phylum: Chordata
Class: Reptilia
Order: Testudines
Family: Cheloniidae
Genus: Chelonia
Species: mydas
Common name: Green turtle

The green turtle was first described by Linnaeus in 1758 and named Testudo mydas, with Ascension Island in the Atlantic as the type locality. Schweigger in 1812 first applied the binomial C. mydas in use today. Overall, the genetic structure of the green turtle rookeries shows distinctive mitochondrial DNA (mtDNA) properties for each nesting region (Bowen et al., 1992). The geographical scale of genetic population structure varies among regions and genetic differentiation is generally detected between rookeries separated by more than 500 km (Dethmers et al., 2006; Jensen et al., 2013). Mitochondrial DNA data suggest that the global matriarchal phylogeny of green turtles has been shaped by ocean basin separations (Bowen et al., 1992; Encalada et al., 1996) and by natal homing behavior (Meylan et al., 1990). Within the eastern Pacific Ocean, specific or subspecific status has been applied to green turtles (also known as black turtles; (C. mydas agassizii) ranging from Baja California south to the Republic of Peru (Peru) and west to the Revillagigedos Islands and Galápagos Archipelago (Márquez-Millán, 1990; Pritchard, 1997); however, genetic analyses do not support such taxonomic distinctiveness (Bowen et al., 1992; Karl et al., 1992). As a result, for the purposes of this Status Review, we consider the global green turtle population to be single species, C. mydas, with 11 distinct population segments as described in Section 4.

2.2. Physical Appearance

The green turtle grows to a maximum size of about 1 m in shell length and a weight of 200 kg. It has a heart-shaped shell, small head, and single-clawed flippers. The carapace has five vertebral scutes, four pairs of costal scutes, and 12 pairs of marginal scutes. The head has a single pair of elongate prefrontal scales, four postorbital scales behind each eye, both of which are distinguishing characteristics that set this species apart from other hard-shell sea turtles. Green turtles have a lower jaw-edge that is coarsely serrated, corresponding to strong grooves and ridges on the inner surface of the upper jaw (Carr, 1952; Pritchard and Trebbau, 1984; Hirth, 1997).

The term “green” refers not to the external coloration, but to the color of the turtle’s subdermal fat. The carapace of adult green turtles is light to dark brown, sometimes shaded with olive, with
radiating wavy or mottled markings of a darker color or with large blotches of dark brown (Carr, 1952). The carapace coloration changes as the turtle grows from a hatchling to an adult. The dorsal coloration of the green turtle likely has adaptive significance as camouflage from chief predators while the turtle rests motionlessly on the bottom amongst coral and other benthic substrate. The adult plastron ranges from yellowish to orange, although in the East Pacific form there is considerable grayish and charcoal pigment. All hatchling green turtles have a black dorsal surface and a pure white ventral surface.

2.3. Distribution

The green turtle has a circumglobal distribution, occurring throughout tropical, subtropical waters, and, to a lesser extent, temperate waters. Their movements within the marine environment are not fully understood, but it is believed that green turtles inhabit coastal waters of over 140 countries (Groombridge and Luxmoore, 1989).

Figure 2.1. Nesting distribution of green turtles around the globe. Sites marked with an ‘X’ indicate known nesting sites, but for which no recent nesting abundance data are available. THE KEY IN BOTTOM RIGHT CORNER IS NOT CLEAR – “X---n/a”. PERHAPS REPLACE n/a WITH “unquant” = unquantified.

The sites included in the figure are those that were considered for the present Status Review Nesting occurs in more than 80 countries worldwide (Hirth, 1997; Figure 2.1). The primary nesting rookeries (i.e., sites with ≥ 500 nesting females per year) are located at Ascension Island (Mortimer and Carr, 1987), Commonwealth of Australia (Australia; eastern: Limpus, 2009; western: Prince, 1993), Comoros Islands (Frazier, 1985), Eparis Islands (Tromelin Island and Europa Island: Le Gall et al., 1986), Federative Republic of Brazil (Brazil; Trindade Island: Moreira et al., 1995), Kingdom of Saudi Arabia (Saudi Arabia; Miller, 1989), Malaysia (de Silva, 1982), Republic of Costa Rica (Costa Rica; Pacific Coast: Blanco et al., 2012; Caribbean coast: Carr et al., 1982; Bjorndal et al., 1999), Republic of Ecuador (Ecuador; Galapagos Archipelago: Green, 1983), Republic of Guinea-Bissau (Guinea-Bissau; Bijagos Archipelago: Barbosa et al., 1998), Republic of Indonesia (Indonesia; Schulz, 1987), Republic of the Philippines (de Silva, 1982), Republic of Seychelles (Seychelles; Mortimer, 1984), Republic of
Lesser nesting areas are located in Atoll of Manuae (Scilly Atoll; Lebeau, 1985), Bolivarian Republic of Venezuela (Venezuela; Medina and Solé as cited in Ogren, 1989), Chagos Archipelago (Mortimer and Day, 1999), Cook Islands (Palmerston Atoll; Powell, 1957), Cooperative Republic of Guyana (Guyana; Pritchard, 1969), Commonwealth of Australia (Australia; Gulf of Carpentaria; Limpus, 2009), Democratic Republic of Yemen (Yemen; Hirth and Carr, 1970), Democratic Socialist Republic of Sri Lanka (Sri Lanka; Dattatri and Samarajiva, 1983), Dominican Republic (Ottenwalder, 1981), d'Entrecasteaux Islands (Pritchard, 1994), Federative Republic of Brazil (Brazil; Atoll da Rocas; Bellini et al., 1996), Federal Republic of Somalia (Somalia; Goodwin, 1971), Guiana (French Guiana; Fretey, 1984), Independent State of Papua New Guinea (Papua New Guinea; Salm, 1984), Islamic Republic of Iran (Iran; Tuck Jr., 1977), Islamic Republic of Pakistan (Pakistan; Kabraji and Firdous, 1984), Japan (Suganuma, 1985), Kingdom of Thailand (Thailand; Groombridge and Luxmoore, 1989), Mayotte Archipelago (Fretey and Fourmy, 1996), Micronesia (Wetherall et al., 1993), Natuna Islands (Limpus, 1985, 2009), New Caledonia (Limpus, 1985, 2009), People's Republic of Bangladesh (Bangladesh; Khan, 1982), People's Republic of China (China; Groombridge and Luxmoore, 1989), Primeras Islands (Hughes, 1974), Republic of Angola (Angola; Carr and Carr, 1991), Republic of Equatorial Guinea (Equatorial Guinea; Bioko Island: Tomás et al., 1999), Republic of Ghana (Ghana; Fretey, 2001), Republic of Cuba (Cuba; Nodarse et al., 2000), Republic of Cyprus (Cyprus; Kasparek et al., 2001), Republic of India (India; Kar and Bhaskar, 1982), Republic of Indonesia (Indonesia; Aru Islands: Dethmers, 2000), Republic of Kenya Wamukoya et al., 1996), Republic of Madagascar (Madagascar; Rakotonirina and Cooke, 1994), Republic of the Maldives (Maldives; Frazier, 1990), Republic of Mauritius (Mauritius; Groombridge and Luxmoore, 1989), Republic of Nicaragua (Nicaragua; P. Torres, Fauna and Flora International, pers. comm, 2013), Republic of Palau (Palau; Palau BMR, 2008; Maison et al., 2010), Republic of Sierra Leone (Sierra Leone; Fretey and Malaussena, 1991), Republic of Taiwan (Taiwan; Chen and Cheng, 1995), Republic of Turkey (Turkey; Kasparek et al., 2001), Republic of the Union of Myanmar (Myanmar; Kar and Bhaskar, 1982), Sao Tome é Principe (Brongersma, 1982), Socialist Republic of Vietnam (Vietnam; Hien, 2002), Solomon Islands (Vaughn, 1981), United Republic of Tanzania (Tanzania; Howell and Mbindo, 1996), and United Mexican States (Mexico; Revillagigedos Islands: Brattstrom, 1982; Awbrey et al., 1984), and sporadic nesting occurs in at least 30 additional countries (Groombridge and Luxmoore, 1989). Detailed information on distribution and habitat by ocean basin follows.
2.4. **Habitat or ecosystem conditions**

Most green turtles spend the majority of their lives in coastal foraging grounds. These areas include fairly shallow waters both open coastline and protected bays and lagoons. While in these areas, green turtles rely on marine algae and seagrass as their primary diet constituents, although some populations also forage heavily on invertebrates. These marine habitats are often highly dynamic and in areas with annual fluctuations in seawater and air temperatures, which can cause the distribution and abundance of potential green turtle food items to vary substantially between seasons and years (Carballo *et al.*, 2002). Many prey species that are abundant during winter and spring periods become patchy during warm summer periods. Some species may altogether vanish during extreme temperatures, such as those that occur during El Niño Southern Oscillation events (Carballo *et al.*, 2002).

Conditions at coastal foraging areas have been shown to impact the timing of green turtle reproduction (Limpus and Nicholls, 1988; Solow *et al.*, 2002). Therefore, despite the fact that foraging areas are usually separated from nesting areas by hundreds to thousands of kilometers, they have a profound influence on population dynamics of green turtles. Annual and decadal oscillations likely play a large role; however, a better understanding is needed concerning how environmental variability triggers or limits green turtle migration and reproduction. In addition, red tide episodes at foraging areas may lead to mortality of juvenile and adult green turtles, thereby impacting a population's present and future reproductive status (L. Sarti, CONANP, pers. comm., 2007; J. Seminoff, NMFS, pers. obs., 2013). **WHAT KIND OF MORTALITY? LARGER RELATIVE TO COLD STUNS? PERHAPS MENTION COLD STUNS HERE IF THEY ARE COMPARABLE**

In addition to coastal foraging areas, oceanic habitats are used by oceanic-stage juveniles, migrating adults, and, on some occasions, by green turtles that reside in the oceanic zone for foraging. Despite these uses of the oceanic zone by green turtles, much remains to be learned about how oceanography affects juvenile survival, adult migration, and prey availability.

At nesting beaches, green turtles rely on safe and healthy beaches characterized by intact dune structures, native vegetation, without artificial lighting, and normal beach temperatures for nesting (Limpus, 1971; Salmon *et al.*, 1992; Witherington, 1997). Coastal areas denuded of vegetation or with coastal construction can impact thermal regimes on beaches and thus affect the incubation and resulting sex ratio of hatching turtles. Nests laid in these areas are at a higher risk due to tidal inundation (Schroeder and Mosier, 2000). Further, climate change may impact these beaches through sea level rise (Baker *et al.*, 2006; IPCC, 2007) and eventually, lethal incubation temperatures on nesting beaches (Glen and Mrosovsky, 2004). **Fuentes et al. 2010, 2011**

2.5. **Biological Characteristics**

2.5.1. **Nesting and Egg Development**

Green turtles nest on sandy, ocean-facing mainland and island beaches (Hirth, 1997). Although specific characteristics vary between rookeries, green turtle nesting beaches tend to have intact
dune structures and native vegetation (Ackerman, 1997). Nests are typically laid at night at the base of the primary dune (Hirth, 1997; Witherington et al., 2006). Sea turtle eggs require a high-humidity substrate that allows for sufficient gas exchange and temperatures conducive to embryo development (Miller et al., 1997, 2003). Mean clutch size varies greatly among green turtle populations, but on average is approximately 100 eggs per clutch (Hirth, 1997). Green turtle nests incubate for variable periods of time, and length of the incubation period is inversely related to nest temperature (Mrosovsky, 1980). Within a biologically-tolerable range of approximately 26–32 °C, the warmer the sand surrounding the egg chamber, the faster the embryos develop (Mrosovsky and Yntema, 1980; Ackerman, 1997). Temperatures prevailing during the middle third of the incubation period also determine the sex of hatchlings (Mrosovsky and Yntema, 1980), with temperatures near the upper end of the tolerable range producing only female hatchlings and those near the lower end of the range producing only males. The pivotal temperature (i.e., the constant incubation temperature that produces equal numbers of males and females) in green turtles greatly varies with population, ranging from approximately 28.0–30.3 °C (summarized by Godfrey and Mrosovsky, 2006).

2.5.2. Life Cycle

Green turtle hatchlings pip and escape from their eggs and then move upward and out of the nest over a period of several days (Hendrickson, 1958; Carr and Ogren, 1960). Hatchlings emerge from their nests en masse almost exclusively at night and presumably use decreasing sand temperature (i.e., nighttime) as a cue (Hendrickson, 1958; Mrosovsky, 1968). Glen et al. (2006) concluded the most likely cue for green turtle hatchling emergence was subsurface sand temperatures (10–20 cm), with emergence inhibited when temperatures were increasing and most emergences occurring during nighttime hours. After an initial emergence, there may be secondary emergences on subsequent nights (Carr and Ogren, 1960; Witherington, 1986).

Immediately after hatchlings emerge from the nest, they begin a period of frenzied activity. During this active period, hatchlings crawl to the surf, swim, and are swept through the surf zone (Carr and Ogren, 1960; Carr, 1961; Wyneken and Salmon, 1992). Hatchlings first use light visual cues, orienting to the brightest horizon, which is over the ocean on naturally-lighted beaches without artificial lighting (Daniel and Smith, 1947; Limpus, 1971; Salmon et al., 1992; Witherington and Martin, 1996; Witherington, 1997; Stewart and Wyneken, 2004). After reaching the surf, hatchlings frenzy swim away from the beach and are swept through the surf zone, after which wave orientation occurs in the nearshore area and later magnetic field orientation as they proceed further toward open water (Lohmann and Lohmann, 2003).

Upon leaving the nesting beach and entering the marine environment post-hatchling green turtles begin an oceanic juvenile phase during which time they are presumed to primarily inhabit areas where surface waters converge to form local downwellings, resulting in linear accumulations of floating material, especially Sargassum sp.. This association with downwellings is well-documented for loggerheads, as well as for some post-hatchling green turtles (Witherington et al., 2006; 2012). The smallest of oceanic green turtles associating with these areas are relatively active, moving both within Sargassum sp. mats and in nearby open water, which may limit the ability of researchers to detect their presence as compared to relatively immobile loggerheads (Smith and Salmon, 2009; Witherington et al., 2012). Food items documented for a limited
number of stranded post-hatchling green turtles have included predominantly *Sargassum* sp. and associated hydroids, bryozoans, polychaetes, gastropods, as well as cnidarians and other pelagic invertebrates, fish eggs, and debris (Witherington *et al.*, 2006; Boyle and Limpus, 2008; Jones and Seminoff, 2013). In the eastern Pacific Ocean, green turtles reportedly forage on a greater proportion of invertebrate foods, with omnivorous diets reported in turtles throughout the region (Seminoff *et al.*, 2003; López-Mendilaharsu *et al.*, 2005; Amorocho and Reina, 2007; Carrión-Cortez *et al.*, 2010; Lemons *et al.*, 2011).

Oceanic-stage juvenile green turtles originating from nesting beaches in the Northwest Atlantic appear to use oceanic developmental habitats and move with the predominant ocean gyres for several years before returning to their neritic foraging and nesting habitats (Musick and Limpus, 1997; Bolten, 2003). For example, green turtles reared for 6–9 months post-hatching and subsequently released into the Gulf Stream off of Florida, U.S., initially traveled along the current, with some following it to mesoscale eddies of the Northwest Atlantic and others moving back into the nearshore neritic zone prior to returning to the Gulf Stream (Mansfield and Wyneken, 2013). Larger neonate green turtles (at least 15–26 cm straight carapace length; SCL) are known to occupy *Sargassum* sp. habitats and surrounding epipelagic waters, where food items include *Sargassum* sp. and associated invertebrates, fish eggs, insects, and debris (Witherington *et al.*, 2012). Knowledge of the diet and behavior of the oceanic stage, however, is very limited.

The neritic juvenile stage begins when green turtles exit the oceanic zone and enter the neritic zone (Bolten, 2003). The age at recruitment to the neritic zone likely varies with individuals leaving the oceanic zone over a wide size range (summarized in Avens and Snover, 2013). Using skeletochronology, Goshe *et al.* (2010) estimated the duration of the oceanic juvenile stage to be between 1 and 7 years (mean=3, SD=1.6) in the northwestern Atlantic, with juveniles recruiting to neritic habitats over a size range of 19–30 cm SCL (Mendonça, 1981; Goshe *et al.*, 2010). Earlier skeletochronology studies estimated the age of neritic green turtles in the smallest size classes as 3–5 years in Florida (25–35cm SCL; Zug and Glor, 1998) and 4–10 years in Hawaii (36–37cm SCL; Zug *et al.*, 2002). Age and size at recruitment have been estimated at 5-6 years and 40 cm curved carapace length (CCL), respectively, for the northern Great Barrier Reef (nGBR; Limpus and Chaloupka, 1997; Chaloupka *et al.*, 2004) and at 35–40 cm in the eastern Pacific Ocean (Seminoff *et al.*, 2003). Stable isotope analysis indicated that new recruits to the Commonwealth of the Bahamas (Bahamas) had previously spent 3–5 years as oceanic carnivores prior to moving to the neritic zone and a primarily herbivorous diet (Reich *et al.*, 2007). Diet analysis of bycaught green turtles from the oceanic areas of the Central North Pacific revealed a carnivorous diet for individuals 30-70 cm CCL (Parker *et al.*, 2011).

After migrating to the neritic zone, juveniles continue maturing until they reach adulthood, and some may periodically move between the neritic and oceanic zones (NMFS and USFWS, 2007; Parker *et al.*, 2011). The neritic zone, including both open coastline and protected bays and lagoons, provides important foraging habitat, inter-nesting habitat, and migratory habitat for adult green turtles (Plotkin, 2003; NMFS and FWS, 2007). Some adult females may also periodically move between the neritic and oceanic zones (Plotkin, 2003; Hatase *et al.*, 2006) and, in some instances, adult green turtles may reside in the oceanic zone for foraging (NMFS and USFWS, 2007; Seminoff *et al.*, 2008; Parker *et al.*, 2011). Despite these uses of the oceanic
2.5.3. Diet

Green turtles have been shown to consume a wide variety of seagrass, marine algae, and invertebrates (see Bjorndal, 1997). Limited studies on oceanic adults have shown them to be primarily carnivorous (Arthur et al., 2008; Parker et al., 2011). Neritic stage juvenile and adult green turtles are primarily herbivorous, foraging on seagrasses and/or marine algae, although some populations appear to forage heavily on invertebrates (Bjorndal, 1997; Jones and Seminoff, 2013). Some populations may exhibit one or more ontogenetic shifts in diet after recruitment to the neritic zone (Arthur et al., 2008; Howell et al., 2013). At least one population is known to have integrated invasive plant species into its diet (Russell and Balazs, 2009). Detailed diet characterizations have been conducted for relatively few coastal regions, however, and little information is available about differences or similarities in diet at various life stages.

2.5.4. Demographic Features

The primary demographic features of green turtles that are relevant for interpreting population abundance and long-term trends include age-to-maturity (also known as age at first reproduction), reproductive longevity, reproductive output (i.e., egg production, clutch frequency, internesting interval), and annual survivorship. For a summary of known survivorship values and other demographic parameters of green turtles around the world see Table 2.1.

Most green turtles exhibit particularly slow growth rates, which has been described as a consequence of their largely herbivorous (i.e., low net energy) diet (Bjorndal, 1982). Growth rates of juveniles vary substantially among populations, ranging from <1 cm/year (Green, 1993) to >5 cm/year (Eguchi et al., 2012), likely due to differences in diet quality, duration of foraging season (Chaloupka et al., 2004), and density of turtles in foraging areas (Bjorndal et al., 2000; Seminoff et al., 2003; Balazs and Chaloupka, 2004). In general, there is a tendency for green turtles to exhibit monotonic growth (declining growth rate with size) in the Atlantic and non-monotonic growth (growth spurt in mid size classes) in the Pacific, although this is not always the case (Chaloupka and Musick, 1997; Seminoff et al., 2002c; Balazs and Chaloupka, 2004a; Kubis et al., 2009). Consistent with slow growth, age-to-maturity for the green turtles appears to be the longest of any sea turtle species (Chaloupka and Musick, 1997; Hirth, 1997). Size and age at first reproduction has been estimated for green turtles using several methods, including mark-recapture, skeletochronology, and marked, known-aged individuals. Estimates vary widely among studies and populations, and methods continue to be developed and refined (Avens and Snover, 2013). East Pacific green turtles are known to mature at smaller sizes (60–77 cm SCL; Seminoff et al., 2002) than conspecifics in the Northwestern Atlantic (85–100+ cm SCL), Hawaii (80+ cm SCL), and Australia (95 cm CCL; Avens and Snover, 2013). Published age at sexual maturity estimates are as high as 35–50 years, with lower ranges reported from known age turtles from the Cayman Islands (15–19 years; Bell et al., 2005) and Caribbean Mexico (12–20
years; Zurita et al., 2012) and some mark-recapture projects (e.g., 15–25 years in the Eastern Pacific; Seminoff et al., 2002). Mean adult reproductive lifespan of green turtles from Australia’s southern Great Barrier Reef has been estimated at 19 years using mark-recapture and survival data (Chaloupka and Limpus, 2005). The maximum nesting lifespan observed in a 27-year tag return dataset from Trindade Island, Brazil was 16 years; however, nesting monitoring was discontinuous over time (Almeida et al., 2011). Tag return data comprising 2,077 females (42,928 nesting events, 1968–partial 2012 season) from continuous monitoring at French Frigate Shoals, Hawaii show maximum nesting lifespans of 37–38 years (n=2), with many individuals (n=54) documented nesting over a minimum of 25–35 years (I. Nurzia-Humburg, S. Hargrove, and G. Balazs, NMFS, unpublished data, 2013).

Considering that mean remigration intervals range from 2 to 5 years (see Hirth, 1997 for review), these reproductive longevity estimates suggest that a female may nest 3 to 11 seasons over the course of her life. Based on the reasonable means of 3 nests/season and 100 eggs/nest (Hirth, 1997), a female may deposit 9 to 33 clutches, or about 900–3,300 eggs, during her lifetime. These are very approximate estimates, but they nonetheless provide a basis for characterizing reproductive effort in green turtles.

Survivorship has been quantified for green turtles resident to foraging areas as well as for adult females at nesting beaches. In general, survivorship tends to be lower for juveniles and subadults than for adults. In the southern Great Barrier Reef, Chaloupka and Limpus (2005) provided estimates for mean annual adult survival (0.95) that was significantly higher than survival for either subadults or juveniles (0.85 and 0.88, respectively). Seminoff et al. (2003) reported mean annual survival of adults and juveniles in the Gulf of California as 0.97 and 0.58, respectively. However, Eguchi et al. (2010) found the annual survival rates of both juveniles and adults was 0.86 at a northern foraging ground in the eastern Pacific. At a Bahamas foraging habitat, juvenile green turtle survivorship was considerably higher at 0.89, although this value dropped to 0.76 once turtles emigrated from this protected site (Bjorndal et al., 2003). Low survivorship as a result of human impacts has also been reported for a Caribbean Nicaraguan foraging area where Campbell and Lagueux (2005) found low survival (0.55) among large juveniles and adults; they also report annual survival of adults nesting at Tortuguero of 0.82, which is close to the value of 0.85 reported by Troëng and Chaloupka, 2007 for the same nesting site. Therefore, it is apparent that the survivorship at any particular site will be influenced by the level of human impacts, with the more pristine green turtle stocks tending to represent more 'natural' survivorship values (e.g., Great Barrier Reef) and others with survivorship values largely influenced by anthropogenic impacts (e.g., Nicaragua). IT WOULD BE NICE TO HAVE A SUMMARY TABLE OF SURVIVORSHIP ESTIMATES FROM DIFFERENT POPULATIONS and STUDIES, SIMILAR TO TABLE 2.1.
Table 2.1. Demographic parameters of green turtles at nesting sites worldwide. For a summary of the data sources for each field entry, see Appendix 1.

<table>
<thead>
<tr>
<th>Nesting Site</th>
<th>Mean Nesting Size (SCL, cm)</th>
<th>Remigration Interval (yrs.)</th>
<th>Nesting Frequency (nests/yr)</th>
<th>Clutch Size (eggs/clutch)</th>
<th>Survival Rates</th>
<th>Growth Rates (SCL; cm/year)</th>
<th>Age at First Reproduction (years)</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>NORTH ATLANTIC</strong></td>
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<tr>
<td>Archie Carr National Wildlife Refuge, Florida, USA</td>
<td>101.5</td>
<td>2</td>
<td>3</td>
<td>136</td>
<td>26–75% hatchling</td>
<td>Adult=F: 0.5; hatchling: 1.18-4.55</td>
<td>18–27</td>
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<td>Core Index Beaches, FL, USA</td>
<td>–</td>
<td>–</td>
<td>10 max</td>
<td>128</td>
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<tr>
<td>El Cuyo, Yucatan, Mexico</td>
<td>–</td>
<td>–</td>
<td>2.73</td>
<td>129.7</td>
<td>–</td>
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<tr>
<td>Isla Holbox, Quintana Roo, Mexico</td>
<td>–</td>
<td>2-3</td>
<td>–</td>
<td>113.3</td>
<td>–</td>
<td>–</td>
<td>12–20, mean 16</td>
</tr>
<tr>
<td>Central Coast, Quintana Roo, Mexico</td>
<td>–</td>
<td>2-3</td>
<td>–</td>
<td>116</td>
<td>Hatchling= 81.98%</td>
<td>–</td>
<td>12–20</td>
</tr>
<tr>
<td>Isla Aguada, Campeche, Mexico</td>
<td>–</td>
<td>–</td>
<td>3.54-4.01</td>
<td>112.25</td>
<td>Hatchling= 55.8-61.7%</td>
<td>–</td>
<td>–</td>
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<tr>
<td>Tortuguero, Costa Rica</td>
<td>100.1</td>
<td>2.95</td>
<td>2.8</td>
<td>108</td>
<td>Adult= 55-82%</td>
<td>–</td>
<td>12-26</td>
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<td><strong>MEDITERRANEAN</strong></td>
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<td>Akyantan, Turkey</td>
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<td>Kazanli, Turkey</td>
<td>96</td>
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<td>115</td>
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<td>Samadang, Turkey</td>
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<td>–</td>
<td>125</td>
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<tr>
<td>Alagadi, Cyprus</td>
<td>92 (CCL)</td>
<td>3</td>
<td>3</td>
<td>115</td>
<td>–</td>
<td>0.11 (CCL)</td>
<td>–</td>
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<tr>
<td>West Coast, Cyprus</td>
<td>–</td>
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<td>4–5</td>
<td>100</td>
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<td>Israel</td>
<td>–</td>
<td>3</td>
<td>–</td>
<td>105</td>
<td>–</td>
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</tr>
<tr>
<td>Lattakia Beach, Syrian Arab Republic</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>108</td>
<td>–</td>
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<tr>
<td>Nesting Site</td>
<td>Mean Nesting Size (SCL, cm)</td>
<td>Remigration Interval (yrs.)</td>
<td>Nesting Frequency (nests/yr)</td>
<td>Clutch Size (eggs/clutch)</td>
<td>Survival Rates</td>
<td>Growth Rates (SCL; cm/year)</td>
<td>Age at First Reproduction (years)</td>
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<tr>
<td>Ras al-Bassit, Syrian Arab Republic</td>
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<tr>
<td>Wadi Kandil Beach, Syrian Arab Republic</td>
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<td><strong>SOUTH ATLANTIC</strong></td>
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<td>Bioko Island, Equatorial Guinea</td>
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<td>–</td>
<td>3</td>
<td>104.6–112.4</td>
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<td>Poilão Bijagos</td>
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<td>–</td>
<td>122–131.2</td>
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<td>Archipelago, Guinea Busseau</td>
<td>–</td>
<td>4</td>
<td>3</td>
<td>102.9</td>
<td>–</td>
<td>–</td>
<td>17–35</td>
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<tr>
<td>Aves Island, Venezuela</td>
<td>107.7</td>
<td>2-3</td>
<td>1.6–2.6</td>
<td>122.9–124</td>
<td>–</td>
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<td>–</td>
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<tr>
<td>Galibi Reserve, Suriname</td>
<td>109</td>
<td>2-3</td>
<td>3.5</td>
<td>102–138</td>
<td>–</td>
<td>–</td>
<td>24-36</td>
</tr>
<tr>
<td>Isla Trindade, Brazil</td>
<td>116.8 (CCL)</td>
<td>3</td>
<td>~3–6</td>
<td>120.1</td>
<td>–</td>
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<tr>
<td>Atol De Rocas, Brazil</td>
<td>116.8 (CCL)</td>
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<td><strong>SWUIT IND IN DIAN OCEAN</strong></td>
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<tr>
<td>Aldabra, Seychelles Islands</td>
<td>103</td>
<td>–</td>
<td>3</td>
<td>90–200</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Mohéli, Comoros Islands</td>
<td>112.3 (CCL)</td>
<td>–</td>
<td>–</td>
<td>116</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Mayotte, Comoros Islands, France</td>
<td>110.8 (CCL)</td>
<td>3</td>
<td>3.03</td>
<td>121.6</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Tromelin, Esparces Islands, France</td>
<td>104.1</td>
<td>3</td>
<td>3</td>
<td>124.6–129</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Europa, Esparces Islands, France</td>
<td>108.9</td>
<td>3</td>
<td>3</td>
<td>142–152</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td><strong>NORTH INDIAN OCEAN</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Gujarat, India</td>
<td>–</td>
<td>–</td>
<td>–</td>
<td>92.6</td>
<td>–</td>
<td>–</td>
<td>–</td>
</tr>
<tr>
<td>Nesting Site</td>
<td>Mean Nesting Size (SCL, cm)</td>
<td>Remigration Interval (yrs.)</td>
<td>Nesting Frequency (nests/yr)</td>
<td>Clutch Size (eggs/clutch)</td>
<td>Survival Rates</td>
<td>Growth Rates (SCL; cm/year)</td>
<td>Age at First Reproduction (years)</td>
</tr>
<tr>
<td>--------------------------------------------------</td>
<td>----------------------------</td>
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<td>-----------------------------</td>
<td>--------------------------</td>
<td>----------------</td>
<td>-----------------------------</td>
<td>---------------------------------</td>
</tr>
<tr>
<td>Hawkes Bay and Sandpit, Pakistan</td>
<td>108.5</td>
<td>4</td>
<td>88.5</td>
<td>115</td>
<td>80%</td>
<td>25–30</td>
<td>2</td>
</tr>
<tr>
<td>Sharma, Peoples Democratic Republic of Yemen</td>
<td>96</td>
<td>4</td>
<td>106–122.4</td>
<td>80% (artificial hatcheries)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ras al Hadd, Oman</td>
<td>97.1</td>
<td>4</td>
<td>110.6</td>
<td>80% (artificial hatcheries)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ras Baridi, Saudi Arabia</td>
<td>105.2 (CCL)</td>
<td>4</td>
<td>103</td>
<td>80% (artificial hatcheries)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Karan and Jana Islands, Arabian Gulf, Saudi Arabia</td>
<td>98 (CCL)</td>
<td>4</td>
<td>88.5</td>
<td>80% (artificial hatcheries)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>EAST INDIAN/WEST PACIFIC</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sarawak, Malaysia</td>
<td>104.7</td>
<td>4</td>
<td>91.2–99.0</td>
<td>80% (artificial hatcheries)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Redang Island, Malaysia</td>
<td>100</td>
<td>4</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sipadan, Sabah, Malaysia</td>
<td>87.3</td>
<td>4</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Sabah Turtle Islands, Malaysia</td>
<td>2.39</td>
<td>4.38</td>
<td>91.2–99.0</td>
<td>80% (artificial hatcheries)</td>
<td></td>
<td></td>
<td>25–30</td>
</tr>
<tr>
<td>Berau Islands, Berawan Archipelago, Indonesia</td>
<td>2.9</td>
<td>4.38</td>
<td>91.2–99.0</td>
<td>80% (artificial hatcheries)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Enu Island (Aru Islands), Indonesia</td>
<td>2.8</td>
<td>3.5</td>
<td>115</td>
<td>80% (artificial hatcheries)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Nesting Site</td>
<td>Mean Nesting Size (SCL, cm)</td>
<td>Remigration Interval (yrs.)</td>
<td>Nesting Frequency (nests/yr)</td>
<td>Clutch Size (eggs/clutch)</td>
<td>Survival Rates</td>
<td>Growth Rates (SCL; cm/year)</td>
<td>Age at First Reproduction (years)</td>
</tr>
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<td>----------------------------------</td>
</tr>
<tr>
<td>Turtle Islands, Philippines</td>
<td>99.5 (CCL)</td>
<td>2.5</td>
<td>5</td>
<td>95.61</td>
<td>1% hatchling</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Lanyu, Taiwan, Provence of China</td>
<td>-</td>
<td>4.3</td>
<td>2.8</td>
<td>105.5</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Wan-an, Taiwan, Provence of China</td>
<td>-</td>
<td>4.6</td>
<td>3.2</td>
<td>104.6</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Thameehla Island, Myanmar</td>
<td>-</td>
<td>1–4</td>
<td>-</td>
<td>93</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Pangumbahan, Java, Indonesia</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>107</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Sukamade, Java, Indonesia</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>2.4</td>
<td>113</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Western Australia</td>
<td>-</td>
<td>2–5</td>
<td>2.93</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Sri Lanka</td>
<td>-</td>
<td>2.5–3.5</td>
<td>4</td>
<td>112.1</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>CENTRAL WEST PACIFIC</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Ogasawara Islands, Japan</td>
<td>-</td>
<td>3.7</td>
<td>4.1</td>
<td>102</td>
<td>-</td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>SOUTHWEST PACIFIC</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Heron Island, southern Great Barrier Reef, Australia</td>
<td>106 (CCL)</td>
<td>5.78</td>
<td>5.06</td>
<td>112.0–115.2</td>
<td>Adults=95% Juv=88%</td>
<td>0.6–2.1 (CCL)</td>
<td>40</td>
</tr>
<tr>
<td>Raine Island, northern Great Barrier Reef, Australia</td>
<td>109 (CCL)</td>
<td>5.35</td>
<td>-</td>
<td>103.9</td>
<td>6.7–99.3%</td>
<td>1.2–4.1</td>
<td>25</td>
</tr>
<tr>
<td>CENTRAL SOUTH PACIFIC</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Rose Atoll, American Samoa</td>
<td>94.7</td>
<td>-</td>
<td>-</td>
<td>-</td>
<td></td>
<td>-</td>
<td>-</td>
</tr>
<tr>
<td>Location</td>
<td>Temperature</td>
<td>Size</td>
<td>Age</td>
<td>Depth</td>
<td>Length</td>
<td>Density</td>
<td>Juvenile</td>
</tr>
<tr>
<td>--------------------------------</td>
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</tr>
<tr>
<td><strong>CENTRAL NORTH PACIFIC</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>French Frigate Shoals, USA</td>
<td>92.2</td>
<td>3-4</td>
<td>4</td>
<td>104</td>
<td>–</td>
<td>1.5–2.5</td>
<td>35–40</td>
</tr>
<tr>
<td><strong>EASTERN PACIFIC</strong></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Michoacán, Mexico</td>
<td>82 (CCL)</td>
<td>1.8–3</td>
<td>3.1</td>
<td>65.1</td>
<td>0.72</td>
<td>9–47</td>
<td>Juv=58%; Adult=85–97%</td>
</tr>
<tr>
<td>Galapagos Islands, Ecuador</td>
<td>81.3</td>
<td>3</td>
<td>1.37</td>
<td>82.9</td>
<td>–</td>
<td>0.11–1.57</td>
<td>–</td>
</tr>
</tbody>
</table>
3. **APPROACH TO STATUS REVIEW**

3.1. **Determination of Distinct Population Segments (DPSs)**

The Status Review Team (SRT) considered a vast array of information in assessing whether there are any green turtle populations segments that satisfy the DPS criteria of being both discrete and significant. In anticipation of a green turtle status review, NMFS contracted in 2011 with two post-doctoral associates to collect and synthesize genetic and demographic information on green turtles worldwide. As a result of this effort, the SRT was presented with and evaluated genetic information that was collected and synthesized by National Research Council (NRC) post-doctoral associate Michael Jensen, in collaboration with SRT member Peter Dutton. The SRT also evaluated demographic information that was collected and synthesized by NRC post-doctoral associate Camryn Allen, in collaboration with SRT Chair Jeffrey Seminoff. This included green turtle nesting information; demography; morphological and behavioral data; movements, as indicated by tagging (flipper and passive integrated transponder (PIT) tags) and satellite telemetry data; and anthropogenic impacts. Also discussed and considered were oceanographic features and geographic barriers.

3.1.1. **Discreteness Criteria**

As noted previously, joint NMFS/FWS policy defines a population to be a DPS if it is both discrete and significant relative to the taxon to which it belongs (FWS and NMFS, 1996, 61 FR 4722). Under the policy, a population may be considered discrete if it satisfies either one of the following conditions: (1) it is markedly separated from other populations of the same taxon as a consequence of physical, physiological, ecological, or behavioral factors; or (2) it is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of Section 4(a)(1)(D) of the ESA. According to the policy, quantitative measures of genetic or morphological discontinuity can be used to provide evidence for item (1).

Data relevant to the discreteness question include physical, physiological, ecological, behavioral, and genetic data. Each type of information has strengths as well as weaknesses for drawing inferences about discreteness. Physical features of the habitat, such as current patterns and intervening land masses, can strongly affect rates of migration and connectivity among subpopulations. On the other hand, features that we perceive as likely barriers might not actually restrict movements of the focal species, and vice versa. Tagging of turtles can provide valuable information about movement of individuals, but generally this approach does not indicate whether these movements led to interbreeding or gene flow. Molecular genetic data are useful in this regard because they integrate information about the strength of genetic connectivity over long periods of time. However, molecular markers become more or less homogenous among populations at levels of interbreeding that are fairly low in demographic terms, so it is often difficult to make inferences about demographic independence from genetic data alone.

After lengthy discussion we compiled a list of attributes that suggested various population groups might be considered discrete. We also discussed alternative scenarios for lumping or splitting these potentially discrete units. Each member of the SRT was then given 100 points that
could be distributed among two categories: 1) The unit under consideration is discrete, and 2) the unit under consideration is not discrete. The spread of points reflects the level of certainty of the SRT surrounding a decision to call the unit discrete. For example, if a member were very certain, they might put 95 points in the affirmative category and the other 5 points in the negative category, while if they were not as confident but lean toward considering the unit discrete, they might split the points 60 affirmative and 40 negative. Using this process, the SRT identified 11 population units that received a mean of between 70 and 96.5 affirmative votes for discreteness (Table 4.1 in Section 4.1.1.5), and each of these was evaluated for significance using a similar process based on considerations outlined in the next section.

3.1.2. Significance Criteria

In accordance with the DPS Policy, the SRT next reviewed whether the population segments identified in the discreteness analysis were biologically and ecologically significant to the taxon to which they belong, which in this case is the taxonomic species *C. mydas*. Data relevant to the significance question include the morphological, ecological, behavioral, and genetic data. The SRT considered the following factors, listed in the DPS Policy, in determining whether the discrete population segments were significant:

1. persistence of the discrete segment in an ecological setting unusual or unique for the taxon;
2. evidence that loss of the discrete segment would result in a significant gap in the range of the taxon;
3. evidence that the discrete segment represents the only surviving natural occurrence of a taxon that may be more abundant elsewhere as an introduced population outside its historical range; and
4. evidence that the discrete segment differs markedly from other populations of the species in its genetic characteristics.

A discrete population segment needs to satisfy only one of these criteria to be considered significant, and the joint policy also allows for consideration of other factors if they are appropriate to the biology or ecology of the species. Because criterion (3) is not applicable to green turtles, the SRT focused on criteria (1), (2) and (4). Genetic information can be used to satisfy both criterion (4) for significance and criterion (1) for discreteness, but the joint policy does not indicate whether the same genetic data can be used for both discreteness and significance. In many previous status reviews that use the joint policy (e.g., Pacific herring (Gustafson et al. 2006); southern resident killer whales (Krahn et al. 2004)), presumably neutral molecular markers were used to evaluate discreteness, while the significance evaluations focused more on adaptive genetic differences. Because the latter are difficult to study with molecular markers, those Biological Review Teams (equivalent to a SRT) considered information on life history and behavior that might suggest different adaptations. Occurrence in an unusual ecological setting is also important in this regard, as unusual ecological / environmental features create novel selective regimes that are likely to promote local adaptations.

The SRT listed the attributes that would make potential DPSs (determined to be discrete in the previous step) significant. As in the vote for discreteness, members of the SRT were then given
100 points with which to vote for whether each unit met the significance criteria in the joint policy. Units that had been identified as discrete received a mean of between 70 and 96.5 affirmative points for significance; see Table 4.2 in Section 4.1.1.5).

The SRT thus determined that each of the 11 population units that were identified as distinct were also biologically and ecologically significant, and hence satisfy the criteria for being considered a DPS. They each represent a large portion of the species’ range, and their loss would result in a significant gap in distribution of the species. Further, each unit is genetically unique, with the potential loss of any one representing a significant loss of genetic diversity. Some units represent a unique or unusual ecological setting influenced by local ecological and physical factors, some unique morphological or other demographic characteristics, and others unique movement patterns. See Section 4.2 for a summary of significance considerations for each unit.

Although DPS is a legal term and the SRT recognizes that these population segments are not technically DPSs until or unless they are designated as such in a rulemaking process, for lack of a better term, we refer to these units as DPSs throughout the remainder of the report.

3.2. Characterization of Status and Trends

Complete population abundance estimates do not exist for the 11 DPSs. The data used in this status review represent the best scientific information available, though the data were more robust in some areas than in others. Within the global range of the species, and within each DPS, the primary data available are collected on nesting beaches, either as counts of nests or counts of nesting females, or a combination of both (either direct or extrapolated). Information on abundance and trends away from the nesting beaches is limited and often nonexistent, primarily because these data are, relative to nesting beach studies, logistically difficult and expensive to obtain. Therefore, the primary information source for directly evaluating status and trends of the DPSs was nesting data.

Nesting female abundance estimates for each nesting site or nesting beach is presented in tables in each of the respective DPS sections later in this Document (Sections 5–15). These tables present each nesting aggregation by Country, Nesting Site, Monitoring Period (Years), and Estimated Nester Abundance, taken as the total number of reproductive females that use any given nesting site over time. (Note: this is not the same as annual nester counts). Abundance was estimated using the best scientific information available. When counts of nesters were not available, remigration intervals and clutch frequencies were used to estimate total nester abundance using the following equation:

\[
\text{Adult Female Abundance} = \frac{\text{nests}}{\text{clutch frequency}} \times \text{remigration interval}
\]

Nester abundance distribution is also presented in Green Turtle Nester Abundance Tables. The tables within each DPS Section, but in this case the number of nesters are present within abundance categories (0–10, 11–50, 50–100 etc.), which depict at a glance the number of nesting beaches within each size category. Nesting female abundance for all DPSs is compiled in Table 16.1.

Accompanying the information in the aforementioned tables are bar plots and Population Viability Analysis (PVA) models for the 33 sites with recent, long-term data sets. The bar plots

Comment [A7]: Comment: I didn’t entirely understand how nesting female data are used to get the total number of reproductive female estimates. I think that you take the running three year sum to do this, but this didn’t seem to be spelled out. It was also not clear how you used time series with missing years of data. I would add these details so that the methods are more clear. This is particularly important, as with your simulation approach, you need one-year lambda estimates for each transition, so how you filled gaps is important to understand.

Response: The commenter did not see the individual chapters, so didn’t have the benefit of the equation used to arrive at estimated nester abundances. We have input it here, so it’s clear to all when reading this section.
present abundance over time, indicating trends for beaches where relatively robust information is available. Bar plots representing annual nesting trends were established only for sites that had data collection with consistent protocols and effort for at least 10 years, and with the most recent year of data <10 yrs old (i.e., 2004 and beyond). Bar plots were completed for 20 nesting sites, based on long-term monitoring data distributed among eight of the 11 DPSs (PVAs were completed for another 13, see below). There are several additional datasets that are greater than 10 years in data collection duration, but they are not included here because it is unlikely that a site with data no more recent than 2003 is reflective of the current situation. In our efforts, strong emphasis was placed on using only the most current data sets.

The SRT undertook quantitative PVA modeling to project adult female abundance at nesting sites for which sufficient data were available. PVAs were conducted for nesting beaches that met the following criteria: 1) A minimum of 15 years of nesting abundance data were collected with consistent effort and standardized protocols (slightly longer than the criterion for bar plots above), 2) the most recent year of data is <10 yrs old (i.e., 2004 and beyond), 3) any gap in data collection does not exceed 3 years, which is the most widely reported remigration interval, 4) the most recent data include at 3 least years of sequential data, and 5) the nesting assemblage has a mean annual nesting level of >10 females. PVAs were conducted for only 13 nesting sites worldwide due to lack of quality long-term time series data: Tortuguero (Costa Rica), Isla Aguada (Mexico), Florida Index Beaches (United States), West Coast Cyprus, Sabah Turtle Islands (Malaysia), Royal Navy Center (Thailand), Redang Terengganu (Malaysia), Thameela Island (Myanmar), Chichijima (Japan), Raine Island (Australia), Heron Island (Australia), East Island, Hawaii (United States), and Colola, Michoacán (Mexico). The dearth of nesting sites that met these criteria underscores the need for greater levels of consistent, long-term monitoring around the world.

Although data types included counts of annual beach crawls, nests, and females, for analysis of population viability relative to an absolute abundance biological reference point we converted data to annual nesting females using values for clutch frequency and nesting success from the same site. When such values were not available, we used information from the nearest nesting beach within the same bioregion for which published estimates were present. If necessary this was extended to the DPS, then ocean basin-level. Preference was given to peer-reviewed published values (over grey literature values).

In these analyses, population growth rates were sampled randomly from the empirical distribution (established with the time series) and used to project future trends with a stochastic exponential growth model (Kendall, 2009; Van Houtan, 2011). We simulated 10,000 runs for each series, and used the distribution of endpoints after 100 model years to characterize the projected abundance. We pooled data if data sets were derived from within the same province or state boundary, or if within the same country, but only after confirming that data were collected with similar protocol and effort. Sites for which data were pooled included Florida (USA) Index Beaches, the west coast of Cyprus, and Sabah Turtle Islands in Malaysia.

PVA efforts provide insights about extinction risk over time. To help interpret this risk, we indicate the probability of green turtle nesting populations declining to two different biological reference points, one using a trend-based and the other an abundance-based threshold. The
trend-based reference point for evaluating population forecasts is half of the last observed abundance value, i.e., a 50 percent decline. Risk is calculated as the percentage of model runs that fall below this reference point after 100 years. This reference point is directly relevant to the Productivity critical assessment element (see discussion on critical assessment elements in Section 3.3), and is meant to highlight the population growth trend and its variability. A similar procedure with a 50 percent decline was previously used in other recent sea turtle risk assessments (Snover and Heppell, 2009; Van Houtan, 2011), so its use here enables some direct comparisons of relative risk across species. Green turtles may have an exceptionally old age at maturity, and hence a long generation time, though estimates vary (Green, 1993; Zug et al., 2002; Bell et al., 2005; Goshe et al., 2010; Richards et al., 2011). Acknowledging that generation length in green turtles is unresolved, the 100-year forecast period can be thought of as roughly equivalent to three generations.

The second reference point for evaluating population forecasts was a total adult female abundance of 300 females. For populations for which the mean remigration interval (RI) is 3 yrs, this value effectively equates to 100 females per year. For populations that have a reported RI value other than three, the RI was rounded down to the nearest integer (adjusted RI) and the ‘current’ number of adult females was taken to be the total number of females nesting over the most recent period equating to adjusted RI interval. This absolute measure is directly relevant to the Abundance critical element (see discussion on elements in Section 3.3). We considered 300 total adult females (e.g., 100 females/yr for population with 3/yr remigration interval) to be a reasonable low threshold below which a green turtle nesting site may be subject to negative 'density dependence' influences (Tiwari et al., 2010). Collectively, these two reference points cover both population trend and absolute abundance—the two major factors that characterize almost all PVAs.

In summary, PVA combined nesting or female counts from adjacent years (the running-sum) and projected forward (i.e., it simply extends recent growth trends into the future). The model averages will always approximate monotonic change and will not account for empirical periodicity or project future oscillations; if there is a single trend in empirical observations, it will just carry it forward irrespective of any other potential factors important to population dynamics. Thus PVA modeling has important limitations, and does not fully incorporate other key elements critical to the decision making process such as Spatial Structure or Five-Factors/Threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone.

3.3. Assessment of Extinction Risk

In order to assess extinction risk in a transparent, repeatable way for DPSs with highly variable data, both in terms of quality and length of data sets, the SRT developed and implemented a structured decision making process. This process incorporated empirical data, quantitative models, qualitative data, and expert opinion. We also looked to the means used by other SRTs to guide us. This included the Viable Salmon Population framework laid out in McElhany et al., 2000 and various other Status Review Reports, including those for West Coast Salmon and Steelhead (NMFS, 2003), Hawaiian Insular False Killer Whales (Oleson et al., 2010), 82 species of Corals (Brainard et al., 2011), and Scalloped Hammerhead Sharks (NMFS, 2013). The green
turtle SRT decided to treat the uncertainty explicitly by using a point system to weigh various extinction risk categories, after taking into account all available data. The spread of points among categories for each SRT member is reflective of uncertainty.

The SRT used six critical assessment elements to characterize extinction risk or long-term viability. The first four critical assessment elements were taken directly from McElhany et al. (2000) to determine viability of salmonids: 1) Abundance, 2) Growth Rate or Productivity (which we often refer to as ‘trends’ in this document), 3) Spatial Structure, and 4) Diversity / Resilience. These were chosen because they are reasonable predictors of extinction risk (viability), they reflect general processes that are important to all populations of all species, and they are (at least in theory) quantifiable. The SRT added two more critical assessment elements that the ESA requires us to consider when making determinations about the status of species: (1) threats to the species, or the five factors outlined in Section 4(a)(1) of the ESA, and (2) conservation efforts that are being made on behalf of the species, as outlined in Section 4(b)(1)(A) of the ESA. The SRT decided to assess threats and conservation efforts as part of its extinction risk analysis because predictions about the future persistence of a DPS necessarily include the likelihood that current and future conditions will either promote or threaten its existence. This is particularly important for a long-lived species such as the green turtle that is slow to mature, because effects of ongoing and even some past efforts may not yet be exhibited in the first four population elements.

3.3.1. Population Elements

The SRT used the following guidelines presented in McElhany et al. (2000) when considering the contribution of each of the population elements to the risk of extinction of a given DPS.

With regard to Abundance

- a population should be large enough to have a high probability of surviving environmental variation of the patterns and magnitudes observed in the past and expected in the future;
- a population should have sufficient abundance for compensatory processes to provide resilience to environmental and anthropogenic perturbation;
- a population should be sufficiently large to maintain its genetic diversity over the long term;
- a population should be sufficiently abundant to provide important ecological functions throughout its life-cycle; and
- population status evaluations should take uncertainty regarding abundance into account.

With regard to Growth Rate or Productivity

- a population’s natural productivity should be sufficient to maintain its abundance above the viable levels, even during poor ocean conditions;
- a viable population should not exhibit trends or shifts in traits that portend declines in population growth rate; and
- population status evaluations should take into account uncertainty in estimates of population growth rate and productivity-related elements.
With regard to Spatial Structure
- habitat patches should not be destroyed faster than they are naturally created;
- some habitat patches should be maintained that appear to be suitable or marginally suitable, but currently contain no individuals;
- source subpopulations should be maintained; and
- analyses of population spatial processes should take uncertainty into account.

With regard to Diversity / Resilience
- human-caused factors should not substantially alter variation in traits such as age structure, size, fecundity, morphology, behavior, and molecular genetic characteristics;
- natural processes of dispersal should be maintained;
- human-caused factors should not substantially alter the rate of gene flow among populations;
- natural processes that cause ecological variation should be maintained; and
- population status evaluations should take uncertainty about requisite levels of diversity into account.

3.3.2. Five Factor / Threats Analysis

Under Section 4(a)(1) of the ESA, the Services are required to determine whether any species is an endangered or threatened species because of any of the following factors. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

Because these factors include “other natural or manmade factors affecting its continued existence” (Factor E), they are inclusive of all threats to the species and therefore provide an overview of factors that might have in the past, may in the present, or may in the future negatively affect the species and ultimately cause it to decline. Conducting an analysis serves numerous purposes. It ensures that current or future threats to the species are taken into account when examining the status of a species under the ESA as we must look to the future when determining if a species is in danger of extinction or likely to become so in the foreseeable future. Because sea turtles are a long-lived species that is slow to mature, it is particularly important that we examine ongoing threats and those in the foreseeable future, as these threats may not be reflected in the abundance and trends of nesting females, the measures most often used in population estimates. Identifying the threats to the species or “factors” in its endangerment also provides a link to the cause of the endangerment’s cause(s), and, therefore, information that can be useful in determining measures necessary to conserve the species.

For each DPS, the SRT reviewed both gray and peer-reviewed literature, and relied on member’s knowledge of green turtles and the specific areas they inhabit. To the extent possible, SRT members also contacted experts within their agency and other agencies, organizations and countries, to gather the best available information. Using this information, the SRT evaluated
the potential role that each factor played in a given DPSs abundance trend and the degree to which each factor is likely to affect population growth in the next 100 years. One hundred years was chosen because that time period is often used in projecting out extinction risk for long-lived species, such as in recovery criteria for many large whales and other turtle species (Angliss et al., 2002; NMFS 2005; Conant et al., 2009; NMFS 2010; NMFS 2011). An analysis of the five factors was conducted explicitly for both terrestrial and marine environments. Thorough discussions of the 5-factor analyses are found in the DPS-specific sections (Sections 5-15) of this report.

3.3.3. Conservation Efforts

Section 4(b)(1)(A) of the ESA requires the Services to take into account “those efforts, if any, being made by any State or foreign nation, or any political subdivision of a State or foreign nation, to protect such species...” when making listing decisions. The SRT decided to examine conservation efforts as part of our assessment of extinction risk because such efforts play a vital role in determining the risk of extinction of a DPS. Conservation efforts include actions, activities, or programs designed to eliminate or reduce threats or otherwise improve the status of a species. Some conservation efforts are identified in a conservation agreement or plan, sometimes among various countries such as in the case of a treaty or multinational conservation plan. Among the types of conservation efforts that were identified for green turtles are laws prohibiting direct harvest of turtles and turtle eggs, fishing regulations designed to limit bycatch of turtles, habitat protection of turtle nesting beaches, predator control at beaches, conservation agreements prohibiting take and/or trade of sea turtles, public education efforts, and research and monitoring of turtles.

As was the case with the five factor analysis, the SRT reviewed both gray and peer-reviewed literature for conservation efforts and used their expert knowledge of specific areas as well as that of other green turtle experts within a region. Obtaining current and accurate information about conservation efforts was, in many cases, especially challenging. For instance, identifying treaties, national laws, and local ordinances was often fairly straightforward, but evaluating their effectiveness and whether they’re being implemented or enforced was often more difficult to ascertain. For local conservation efforts, it could be difficult to even identify ongoing efforts, much less how well they are being implemented or their effectiveness. In cases in which nesting and foraging grounds are widespread, it was often extremely difficult or simply was not possible with the assessment process’ time contraints to identify all of the on-going conservation efforts. For this reason, we generally focused our attention on larger-scale, formalized conservation efforts. An evaluation of conservation efforts was made, based on the effectiveness of the conservation effort and the certainty that the effort will be effective. Thorough discussions of conservation efforts for each DPS are found in the DPS-specific sections (Sections 5-15) of this report.

3.3.4. Voting Process

Two or more members of the SRT took the lead in compiling all available information on the six critical assessment elements for each DPS. This information was shared among SRT members several days before a DPS was to be discussed and voted upon. This process followed two 3-day
meetings at the NMFS-Southwest Fisheries Science Center (14–16 November 2012, 5–7 March 2013) in which the proposed DPSs were determined, and everyone was familiarized with each DPS. The SRT member who was the lead for a given DPS gave a short (<45 min) presentation via conference call, and SRT members discussed the DPS, ensuring all questions were answered and comments expressed. In the 24 hours subsequent to the call, DPS leads updated and shared the information on the critical assessment elements to reflect any refinements in information or additional information that was discussed. If a question or concern could not be addressed on the call or in a follow-up e-mail, the vote on that DPS was delayed. This feedback mechanism was deemed vital to the process so as to capture the collective expertise of the entire SRT for any given DPS. After the critical assessment information was updated, voting sheets and instructions were sent to each SRT member within 24 hours by SRT Chair Jeffrey Seminoff. Votes from each SRT member were then submitted to non-voting SRT member Camryn Allen, usually within 48 hour after completion of the conference call deliberation for that respective DPS. Votes were then organized reported anonymously. Separate votes were taken on each of the six critical assessment elements, as well as the overall extinction risk of the DPS, as outlined below.

3.3.4.1. Voting on Critical Assessment Elements

SRT members ranked the importance of each of the four population elements by assigning them a value from 1 to 5 for each DPS, with 1 (very low risk) indicating that it is “unlikely that this element contributes significantly to risk of extinction, either by itself or in combination with other factors” and 5 (very high risk) indicating that “this factor by itself indicates danger of extinction in the near future.”

SRT members ranked the influence of the five factors (threats) on the status of each DPS by assigning a value of 0 (neutral effect on status—this could mean that threats are not sufficient to appreciably affect the status of the DPS, or that threats are already reflected in the population elements), –1 (threats described in the 5-factor analysis suggest that the DPS will experience some decline (<5%) in abundance within 100 yrs), or –2 (threats described in the 5-factor analysis suggest that the DPS will experience significant decline (≥5%) in abundance within 100 yrs).

SRT members ranked the influence of conservation efforts on the status of each DPS by assigning a value of 0 (neutral effect on status—this could mean that conservation efforts are not sufficient to appreciably affect the status of the DPS, or that conservation efforts are already reflected in the population elements), +1 (activities described in Conservation Efforts suggest that the DPS will experience some increase (<5%) in abundance within 100 yrs.), or +2 (activities described in Conservation Efforts suggest that DPS will experience significant increase (≥5%) in abundance within 100 yrs).

The SRT did note in discussions, just as McElhany et al. (2000) noted in their paper about the four population elements, that none of these elements is entirely independent. Indeed, McElhany et al. (2000) noted that “the value ranges for population growth rate considered necessary for a viable population clearly depend on the population’s abundance.” We further note that past threats and, conversely, conservation efforts, clearly affect abundance, growth rates, spatial structure and diversity / resilience. In this case, in order to minimize “double counting” the
consideration of threats and conservation measures, we only considered those that are unlikely to be reflected in data for the population elements. For instance, if a conservation measure has been underway for several decades and is believed to be reflected in population elements, it would not be considered as affecting future population elements, i.e., its effect would be neutral (unless it is likely to increase or decrease in magnitude and therefore effect in the future). Conversely, if a conservation measure has been underway for 5 or 10 years, even if it is believed to be effective, it is unlikely to be reflected in the abundance or trends of nesting females (which is nearly always the value used for these population elements) because green turtles take over 10 years to mature. Therefore, it would be considered separately as to its possible effects on future extinction risk.

We did not attempt to use the values applied to each element by each SRT member to arrive at extinction risk. Rather, we used the votes for each element as an indicator of which weighed more heavily in the overall extinction risk. We then undertook an entirely separate vote for overall extinction risk. Explicitly voting on each element before voting on an overall extinction risk ensured that everyone considered each element in their thought process while contemplating risk thresholds, and helped to make transparent why each member voted as they did in the overall ranking (see below). In a further effort at transparency and to ensure that each member’s vote was interpreted accurately, each SRT member was required to write a short (1-3 sentence) synopsis of why they voted the way they did in the overall extinction risk. These 'voting justifications' are summarized in the Assessment of Extinction Risk subsections each DPS-specific section (Sections 5–15).

3.3.4.2. Voting on Overall Status of the DPS

SRT members provided their expert opinion (via vote) on the likelihood that each DPS would reach critical risk thresholds within 100 years. For purposes of this exercise, the SRT agreed to define critical risk threshold as follows: “A DPS that has reached a critical risk threshold has such low abundance, declining trends, limited distribution or diversity, and/or significant threats (untempered by significant conservation efforts) that the DPS would be at very high risk of extinction with little chance for recovery.”

Each member was given 100 points to spread across risk categories, reflecting their interpretation of the information for that DPS. The spread of points is meant to reflect the amount of uncertainty in the risk threshold bins. Risk categories were <1%, 1-5%, 6-10%, 11-20%, 21-50%, and >50%. Risk categories were chosen to be most meaningful for interpreting whether or how to list the DPSs under the ESA, e.g., there was no point in further delineating values greater than a 50% risk, which is an extremely high risk of extinction (or critical risk threshold), but it seemed worthwhile to differentiate between a DPS that had less than 1% risk of extinction and one that had a 5% risk of extinction. See Appendix 2 for example of voting form.

Population viability and risk assessment analyses are most meaningful when applied to demographically-independent units, such as sea turtle nesting sites where there is little genetic exchange. Because green turtle DPSs include multiple nesting sites it was necessary to find a way to integrate results for the multiple sites into an overall risk assessment for each DPS. In
doing this, the SRT relied heavily on the critical assessment elements of Spatial Structure and Diversity / Resilience, as reflected in the distribution of nesting abundance, within-DPS genetic structure, and within-DPS satellite-tracked movements. All else being equal, DPSs with a number of relatively large populations with stable or increasing growth rates, distributed throughout the geographic range of the DPS, were considered to have higher viability, while those with fewer robust populations, or with robust populations all concentrated in a small geographic area where they might be susceptible to correlated catastrophes, were considered to be at higher risk. Any DPS with low phenotypic and/or habitat diversity were also considered to be at higher risk because the entire DPS could be vulnerable to persistent environmental conditions (Limpus and Nicholls, 1998; Saba et al., 2008; Van Houtan and Halley, 2011) or stochastic catastrophic events (Hawkes et al., 2007; Van Houtan and Bass, 2007; Fuentes et al., 2011). For examples of similar evaluations conducted as part of ESA recovery planning for listed salmon DPSs, see Wainwright et al., 2008; Sands et al., 2009; and Waples et al., 2010.

In assessing viability, the SRT also carefully considered the current and projected future status of each DPS within the context of historical conditions. In identifying green turtle DPSs, the SRT focused on evidence that these population units had been strongly isolated for long enough to develop significant genetic differences. ‘Historical conditions’ refers to the collective factors that allowed the persistence of these population units over evolutionary time scales (which would generally be > 1,000 yrs for a very long-lived species like the green turtle). This does not presume that all rookeries or DPSs were healthy, even historically. Green turtle DPSs may approximate a metapopulation model (e.g., Hanski and Gilpin, 1991) on long time scales, with some populations being sources and other sinks, but each contributing to overall viability and persistence of the DPS.

In this context, the following non-exhaustive examples could be considered as ‘red flags’, or warning signs, of increased extinction risk: (1) a recent loss of significant nesting sites (or spatial portions thereof) without a corresponding increase in other locations, (2) a recent loss of connectivity or increased isolation between nesting sites, (3) a recent significant contraction in the geographic distribution of nesting within the DPS, (4) a recent substantial decline in nesting at important rookeries in the DPS, and/or (5) a recent significant reduction in overall abundance. None of these conclusively demonstrates that a DPS is at high extinction risk; however, the farther the system is away from the historical conditions that were known to be consistent with viability, the more concerned one is that the DPS might not be viable into the future. Conversely, absent compelling evidence to the contrary, a DPS for which recent data suggests all four critical assessment elements are close to what occurred historically would generally be considered to be viable into the future.

3.4. Assessment of Significant Portion of its Range (SPR)

The ESA defines endangered species as “any species which is in danger of extinction throughout all or a significant portion of its range...” and 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.” However, the ESA does not define the terms ‘significant portion of its range’ or ‘foreseeable future.’ The Services have proposed a “Draft Policy on Interpretation of the Phrase ‘Significant Portion of Its Range’” in the Endangered Species Act’s Definitions of
‘Endangered Species’ and ‘Threatened Species’” (76 FR 76987; December 9, 2011). While the policy remains in draft form, the Services are to consider the interpretations and principles contained in the Draft Policy as non-binding guidance in making individual listing determinations, while taking into account the unique circumstances of the species under consideration. The Draft Policy provides that: (1) If a species (the ESA definition of which includes DPSs) is found to be endangered or threatened in a significant portion of its range (SPR), the entire species is listed as endangered or threatened, respectively, and the ESA protections apply across the species’ entire range; (2) a portion of the range of a species is “significant” if the portion’s contribution to the viability of the species is so important that, without that portion, the species would be in danger of extinction; (3) the range of a species is considered to be the general geographical area within which that species can be found at the time FWS or NMFS makes any particular status determination; and (4) if the species is not endangered or threatened throughout all of its range, but it is endangered or threatened within a significant portion of its range, and the population in that SPR is a valid DPS, we will list the DPS rather than the entire taxonomic species (or subspecies).

The SRT reviewed the information on threats and extinction risk to portions of the range for each DPS. The SRT evaluated whether any portion of the range for each DPS, at present, has a substantially higher risk than any other part of the DPS and if these are significant. Only two DPSs were found to potentially have significant portions of their ranges, the Central North Pacific DPS and the East Indian-West Pacific DPS. The SRT conducted two votes for the risk of extinction to these DPSs: one for the entire DPS, and one for the DPS that would remain if the SPR is lost. A summary of the SRT’s discussions and conclusions on SPR for each DPS is found in the DPS-specific sections (Sections 5–15) of this report.

### 3.5 Next Steps

The Status Review process stopped at the assessment of extinction risk described above. The status review will be reviewed by a separate team, which will make the determination of how the species will be listed (number of DPSs) and the status of each of the listed entities under the ESA (threatened, endangered or not warranted/delist). This will be reflected in a proposed rule to make these changes to the current listings of the green turtle.

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1 The draft final policy adds to this statement, “or likely to become endangered in the foreseeable future”, i.e., threatened, but the policy has not been finalized.
Summary of Section 3 Peer Review

I have reviewed Section 3, and also looked through Section 16 to be sure I understood how the results are put together. For the most part, I very much like the approach you have taken. In particular, I thought you do a great job in laying out the reasoning for the different assessment criteria and how they are combined. I do have a few questions and suggestions:

Section 3.1. I don’t have any issues with the determination of the DPSs. The use of multiple criteria is very good and the way you make the determinations seem entirely reasonable.

Section 3.2.

-- I didn’t entirely understand how nesting female data are used to get the total number of reproductive female estimates. I think that you take the running three year sum to do this, but this didn’t seem to be spelled out. It was also not clear how you used time series with missing years of data. I would add these details so that the methods are more clear. This is particularly important, as with your simulation approach, you need one-year lambda estimates for each transition, so how you filled gaps is important to understand.

-- I feel that the simulation approach to predicting probabilities of decline make sense, but I would have liked to see one other analysis to back up the appropriateness of this approach. These simulations assume no trend in the lambda values over time into the future, and hence also over the data collection period. I would therefore have liked to see a simple regression of lambda vs. time period as a way to ask if there is evidence of non-stationarity in growth rates, and in particular if there is evidence of declining mean lambda over time. I realize that long term oceanographic oscillations can mean that such a trend in lambda values might not reflect steady increases or decreases in mean population growth rates into the future (e.g., the van Houtan and Halley paper), but since this is an assumption that your projections are based on (and hence an easy point of attack from someone reviewing the assessment), I would still suggest adding this analysis with a discussion of the difficulty of gauging longer-term stationarity from relatively short time series. You might also note that you are not accounting for the positive autocorrelation in rates that they see, which means your results are likely somewhat optimistic.

-- I like the use of the two different decline thresholds. As there is no single agreed-upon number for a minimum population size, this is a nice approach.

-- I had hoped to look at the actual PVA results (probabilities of reaching either threshold) in table 16.4. The up and down arrows don’t give a sense of what these risk estimates actually are, and I don’t think they were anywhere else either. I also wondered if the up and down arrows are significant upward or downward trends, or something more subjective. I would suggest making these results more explicit. I had hoped to compare these (loosely) with the results of the voting on extinction risk, but couldn’t really do this to see how other factors outweighed or reinforced what these simulation results showed.
Section 3.3. I don’t have any major comments. I like the clear description of the ways that the different criteria were included in the voting. One thing that would be good is to clarify the connection between the first set of votes and the final overall vote on risk – it seems like the first set of votes are essentially to inform the final voting, but I was unsure if there is another purpose as well? Related to this, I was not clear on how the final votes (Tables 16.10 and 16.11) were used to assign the final status of each DPS? This might be clear in sections 5-15, but I would have thought that some clear statement of how the results were used to assign final status would be included in Section 3. This seems especially important to make clear if the goal of Section 3 is to explain the train of logic between the process and the final assessment.

Section 3.4. This section seems fine to me.

I hope that these comments are useful. Please let me know if I can answer any questions to clarify my comments.
4. DETERMINATION OF DPS

4.1. Overview of Information Used to Determine DPS

As noted in Section 1.1.4, joint NMFS/FWS policy provides that a population or group of populations can be considered a “Distinct Population Segment” or DPS if it is both discrete and significant relative to the taxon to which it belongs (which in this case is the taxonomic species C. mydas). The primary criterion for discreteness considered by the SRT was marked separation from other population units within the taxon. To determine whether ‘discrete’ population units were also ‘significant,’ the SRT focused on the following factors: (1) occurrence in an ecological setting unusual or unique for the taxon; (2) whether loss of the discrete segment would result in a significant gap in the range of the taxon; and (3) substantial genetic differences compared to other population segments.

The DPS determinations for green turtles were unusually challenging because they required (1) adopting a global perspective, as this is one of the most widespread and continuously distributed species in the world, and (2) integrating diverse types of information into an overall assessment. Something similar was recently attempted by Wallace et al. (2010a), who collected large amounts of published and unpublished data for C. mydas and integrated this information at different spatial scales. They used the results to identify a total of 17 Regional Management Units (RMUs), which are intended to help guide conservation planning. The SRT found this report to be a very useful reference. However, the criteria used by Wallace et al. (2010a) to identify their RMUs differ from the criteria specified in the joint DPS policy, so it was necessary for the SRT to consider this problem from the perspective of the guidelines provided in the joint policy. The next two sections describe in detail how that was done.

4.1.1. Discreteness Determination

As a first step in evaluating discreteness among the global green turtle population, the SRT began by focusing on the physical separation of ocean basins by continents. The result was an evaluation of data for the three major ocean basins (Pacific Ocean, Atlantic Ocean, and Indian Ocean). This was not to preclude any larger or smaller DPS delineation, but to aid in data organization and assessment. The SRT then evaluated genetic information by ocean basin. The genetic data consisted of results from studies using maternally inherited mitochondrial DNA (mtDNA) and biparentally inherited nuclear DNA (nDNA) microsatellite and single nucleotide polymorphism (SNP) markers. Next, tagging data (both flipper tags and passive integrated transponder (PIT) tags) and telemetry data were reviewed. Additional information, such as potential differences in morphology, was also evaluated. Finally, the SRT considered whether the available information suggests that green turtle population segments are bounded by any oceanographic features (e.g., current systems) or geographic features (e.g., land masses).
Genetics

The green turtle is present in all tropical and temperate ocean basins and has a life history that involves nesting on coastal beaches and foraging in neritic and oceanic habitats, as well as long-distance migrations between and within these areas. As with other globally-distributed marine species, today’s global green turtle population has been shaped by a sequence of isolation events created by tectonic and oceanographic shifts over geologic time scales, the result of which is population substructuring in many areas (Bowen et al., 1992; Bowen and Karl, 2007). Globally, green turtles comprise a mosaic of populations, each with unique nesting sites and in many cases possessing disparate demographic features (e.g., mean body size, age at first reproduction; NMFS and USFWS, 2007). However, despite these differences, green turtles from different populations often mix in common foraging grounds thus creating unique challenges when attempting to delineate distinct population segments for management or listing purposes (Jensen et al., 2013).

Examining the phylogeography of green turtles across their global distribution through mtDNA sequence diversity, Bowen and Karl (2007) found it to be similar to loggerhead turtles (Caretta caretta), with a separation of green turtles in the Atlantic-Mediterranean basins from those in the Indo-Pacific basins dated to the Pleistocene period. Estimates of divergence time between these two primary evolutionary lineages range from approximately 3 million years ago (mya; based on mtDNA data) to 7 mya (based on nDNA data) (Bowen et al., 1992; Dutton et al., 1996; Naro-Maciel et al., 2008; Duchene et al., 2012). This divergence between Pacific and Atlantic evolutionary lineages of Chelonia mydas is thought to have occurred after the cooling of southern ocean waters in the mid to late Miocene (between 17 and 6 mya; Rögl, 1998), and possibly coincided with the closing of the Isthmus of Panama (between 5 and 2.5 mya; Leigh et al., 2013).

Geography and climate appear to have shaped the subsequent evolution of these two matriarchal lineages with the onset of glacial cycles, the appearance of the Isthmus of Panama creating a land barrier between the Atlantic and eastern Pacific (between 5 and 2.5 mya; Farrell et al., 1995), and upwelling of cold water off southern Africa creating an oceanographic barrier between the Atlantic and Indian Ocean (Bowen, 2003). Recent warm temperatures during interglacial periods allowed a reverse invasion from the Atlantic and back into the Indian Ocean although the scale and timing of this connectivity remains unknown (Formia et al., 2006; Bourjea, et al., 2007; Bowen and Karl, 2007). Today it appears that green turtles within a basin appear to be effectively isolated from populations in the other basins.

Mitochondrial DNA studies indicate that regional turtle nesting sites within an ocean basin have been strongly isolated from one another over ecological timescales (Bowen et al., 1992; Bowen and Karl, 2007). These same data indicate strong female natal homing and suggest that each regional nesting population is an independent demographic unit (Bowen and Karl, 2007). It is difficult to determine the precise boundaries of these demographically-independent populations in regions such as Southeast Asia where nesting sites are closely scattered or where they range along large areas of a continental coastline (e.g., Western Australia; Dethmers et al., 2006). There appears to be varying levels of connectivity between proximate nesting sites facilitated by imprecise natal homing and male-mediated gene flow (Bowen et al., 1992; Karl et al., 1992;
Pearse et al., 2001; Dethmers et al., 2006). However, regional genetic populations often are characterized by allelic frequency differences rather than fixed genetic differences. There is concern that current analytical tools are unable to identify discrete or demographically-independent populations based on genetic data when the allelic frequency differences are slight, and when the molecular markers are not sensitive enough to detect structure when it exists (Taylor and Dizon, 1999; Dutton et al., 2013). Recent studies using informative nuclear microsatellite and single nucleotide polymorphism markers have detected greater levels of structuring between nesting sites, and challenging results from earlier studies on the degree to which male-mediated gene flow occurs between regional nesting sites occurs (Dutton et al., 2013; Roden et al., 2013).

Nevertheless, mtDNA studies have shown that high levels of genetic diversity and phylogeographic structure are found in both the Indo-Pacific and the Atlantic and Mediterranean basins. Among 27 green turtle nesting sites in the Indo-Pacific, 25 haplotypes have been observed, with sequence divergences of up to 8.4% (Dethmers et al., 2006). Among the Atlantic and Mediterranean nesting sites sequence divergence is lower, but haplotype diversity is high. A total of 47 Atlantic and Mediterranean haplotypes have been published (Allard et al., 1994; Lahanas et al., 1994; Encalada et al., 1996; Bass and Witzell, 2000; Bass et al., 2006; Bjorndal et al., 2006; Formia et al., 2006, 2007; Naro-Maciel et al., 2007; Foley et al., 2007; Ruiz-Urquiola et al., 2010; Bagda et al., 2012) and at least another 18 haplotypes are yet to be published (http://accstr.ufl.edu/). Extensive mtDNA studies show that the central and eastern Pacific green turtle populations are completely isolated reproductively from the western Pacific/Indian Ocean populations, although foraging ground studies using mixed-stock analysis suggest that at least juveniles from these distinct genetic regions do occasionally disperse across the Pacific (Hamabata et al., 2009; Amorocho et al., 2012; Godoy et al., 2012).

A global phylogenetic analysis based on sequence data from a total of 129 mtDNA haplotypes (from approximately 4,400 individuals sampled from 105 nesting sites) available for green turtle nesting populations around the world was conducted for the SRT (Jensen and Dutton, NMFS, unpublished data; M. Jensen, NRC, pers. comm., 2013). Results indicated that the mtDNA variation present in green turtles throughout the world today occurs within eight major clades that are structured geographically within ocean basins (Figure 4.1). These clades represent relationships (similarities) between haplotypes on evolutionary timescales as opposed to ecological timescales, and would qualify as Evolutionary Significant Units (ESUs) as defined by Moritz (1994).

Comment [A2]: Incomplete citation. Is this Moritz 1994?
Response: Yes…corrected in text
Figure 4.1. **Bayesian** phylogenetic tree showing relationships (average number of base substitutions) among 129 mtDNA haplotypes that group into eight major clades, each defined by a color. The flatback turtle, *Natator depressus* is used as an outgroup. The geographic distribution of haplotypes of the same clade is shown by pie charts with corresponding colors. Each pie chart corresponds to a genetically distinct management unit as described by Moritz (1994; Jensen and Dutton, NMFS, unpublished data). The samples to the right of the tree (in grey/white) are from Saudi Arabia and contain two highly divergent groups of haplotypes. More sampling is needed from this region to assess their placement in the tree.

There is divergence among individual haplotypes within each green turtle clade (M. Jensen, NRC, pers. comm., 2013) and nesting populations within a region are often sub-divided into distinct populations containing no overlap of haplotypes, even though these haplotypes belong to the same broader evolutionary clade (Figure 4.2). One or more hierarchical levels of this genetic diversity might correspond to management units (representing nesting stocks) as defined by Moritz (1994). Two of the eight major mtDNA clades are found in the Atlantic/Mediterranean region. One clade includes populations from the Mediterranean and the western North Atlantic. Within that clade, two strongly divergent groups of haplotypes are found (Figure 4.1), with one group being restricted to the Mediterranean and the other being restricted to the western North Atlantic (Figure 4.2). These two geographically-separated groups of divergent haplotypes thus provide strong evidence for discreteness. The second clade, which includes all remaining Atlantic populations, also includes two different groups of haplotypes: one is found primarily in the eastern Caribbean and northeast coast of South America, and the other occurs only along...
coasts of east central South America and west central Africa, and on islands in the South Atlantic (Figure 4.2). Foraging ground studies in the Atlantic have generally shown regional structuring with strong stock contribution from nearby regional rookeries, but little mixing over long distances (Bolker et al. 2007). In the Southwest Atlantic, foraging areas in Brazil are mainly made up of turtles from Ascension Island, Trinidad and Aves and Surinam (Proietti et al. 2009, 2012). Because of the overlap in haplotype frequencies across nesting populations in the South Atlantic there is insufficient resolution in the genetic data to determine if there is any dispersal across the South Atlantic (Naro-Maciel et al., 2012). Overall, the distribution of the two genetic haplotype lineages (clade I and clade II) is very similar to what is seen for the nesting population and indicates a strong regional structuring with little overlap (Bolker et al., 2007). However, a recent study showed that a large proportion of juvenile green turtles in Cape Verde in the eastern Atlantic originated from distant rookeries across the Atlantic, namely Suriname (38% percent), Ascension Island (12% percent) and Guinea Bissau (19% percent) suggesting that, like loggerheads, green turtles in the Atlantic undertake transoceanic developmental migrations (Monzón-Argüello et al., 2010). The fact that long distance dispersal is only seen for juvenile turtles suggests that larger adult-sized turtles return to forage within the region of their natal rookeries thereby limiting the potential for gene-flow across larger scales (Monzón-Argüello et al., 2010).
Figure 4.2. Haplotype (mtDNA) frequencies (colors) at nesting sites in the Mediterranean and Atlantic (M. Jensen, NRC, pers. comm., 2013). Note that the dominant haplotypes in eastern Caribbean nesting sites (green) are distinct from those in other regions, but are most closely related to haplotypes in the east and central South Atlantic (blue/black) that together are part of one clade (Figure 4.1).

Phylogenetic analysis of mtDNA sequence data indicate that the different nesting sites cluster into the equivalent geographic groups (Figure 4.3). The presence of haplotypes from both of the divergent clades in the central Caribbean is believed to be the result of multiple colonization events over evolutionary time, involving range expansion and contraction from ancestral populations in the central, eastern and southern Atlantic (see Shamblin et al., 2012).
In the southwest Indian Ocean, Bourjea et al. (2007) used 396 base pairs (bp) of the mtDNA control region to assess the population structure among 288 nesting green turtles from 10 nesting sites. They identified seven haplotypes (Figure 4.4). Overall, the southwest Indian Ocean appears to have at least two genetic stocks: 1) the South Mozambique Channel consisting of Juan de Nova and Europa, and 2) the numerous nesting sites in the North Mozambique Channel consisting of Nosy Iranja, Mayotte, Mohéli, Glorieuses, Cosmoledo, Aldabra, Farquhar, also including Tromelin located east of the Republic of Madagascar (Madagascar; Figure 4.5).
Figure 4.4. Haplotype (mtDNA) frequencies (colors) at sampled nesting sites in the Indian Ocean (M. Jensen, NRC, pers. comm., 2013).
Figure 4.5. Phylogenetic Genetic groupings (Neighbor-Joining tree of FST Values) among green turtle nesting sites in the Indo-Pacific. The Indo-Pacific includes the southwest Indian Ocean (SWIO), northwest Indian Ocean (NWIO) and east Indian and western Pacific Oceans (EIO/WP). Groupings are based on 396 bp mtDNA sequence data (M. Jensen, NRC, pers. comm., 2013).

Bourjea et al. (2007) suggest that the South Mozambique Channel could be further subdivided in two different genetic stocks, one in Europa and the other one in Juan de Nova based on a significant haplotype frequency shift. Interestingly, they recorded a high presence of an Atlantic Ocean haplotype (CM-A8) in the two most southern nesting sites, Juan de Nova and Europa. CM-A8 is common and widespread across the South Atlantic and appears to be the ancestral haplotype in the South Atlantic. This suggests that gene flow has occurred from the Atlantic Ocean into the Indian Ocean via the Cape of Good Hope, but it remains unclear if this is a rare dispersal event. The Northern nesting sites on the other hand share several haplotypes (including CmP47 and CmP49) with nesting sites in the eastern Indian Ocean, Southeast Asia and the Western Pacific (Figure 4.4) indicating strong connectivity with the eastern Indian Ocean populations. There is also evidence of connectivity with the Australian Great Barrier Reef (GBR), however, it is not known whether this is the result of contemporary gene flow or multiple rare historical colonization events.

The southwest Indian Ocean has a mix of common and widespread haplotypes, indicating that this is a region of high genetic diversity, with 0.3–6.5 percent (mean=4.2
percent) estimated sequence divergence among the seven haplotypes identified. These haplotypes belong to three highly diverged genetic clades and highlight the complex colonization history of the region (Figure 4.1). There are no nuclear DNA studies from this region.

Very limited information from only a single nesting site (Jana Island, Saudi Arabia, n=27) exists on the genetic structure from the north Indian Ocean (M. Jensen, NRC, pers. comm., 2013). Four mtDNA haplotypes never reported from any other nesting site have been identified from Jana Island, and are highly divergent from other haplotypes in the Indian Ocean (Figure 4.1). Despite limited sampling from this region, it is clear that this nesting site is isolated from those in the south Indian Ocean and western Indian Ocean. However, more sampling is needed to resolve the number of genetic stocks.

Genetic sampling in the east Indian and western Pacific Ocean regions (EIO/WP) has been extensive with more than 22 nesting sites sampled. However, there are a high number of nesting sites in this region, there is complex structure, and there are gaps in sampling relative to distribution (e.g., Thailand, Vietnam, parts of Indonesia, and the Philippines). Overall, nesting populations in this region have varying levels of spatial structure characterized by a few common and widespread haplotypes. Most genetic stocks are identified by frequency shifts of common haplotypes supported by the presence of rare or unique haplotypes.

Significant population substructuring (pairwise $F_{ST}$ 0.10-0.95, $p<0.05$) occurs among nesting sites in the EIO/WP. Of 26 nesting sites studied, 18 regional genetic stocks have been identified in the EIO/WP: Northwest Shelf, Scott Reef, Ashmore Reef, the Gulf of Carpentaria, Cocos “Keeling” Island, and Cobourg Peninsula (Dethmers et al., 2006; Jensen, 2010; Commonwealth of Australia (Australia)), West Java, Berau Islands, and Aru (Indonesia), Peninsular Malaysia, Sarawak, and Southeast Sabah (Malaysia), Sulu Sea (Malaysia/Philippines; Dethmers et al., 2006), Wan-an Island, and Lanyu Island (Taiwan; Cheng et al., 2008), Zamami Island (Hamabata et al., 2009), Iriomote Island, and Ishigaki Island (Japan; Nishizawa et al., 2011; Figure 4.4 and 4.5).

Mixed-stock analysis of foraging grounds show that green turtles from multiple nesting beaches commonly mix at feeding areas across northern Australia (Dethmers et al., 2006) and Malaysia (Jensen, 2010), with higher contributions from nearby large nesting sites.

Genetic sampling in the southwest Pacific has been extensive for larger nesting sites along the GBR, the Coral Sea and New Caledonia (Dethmers et al., 2006; Jensen, 2010; P. Dutton, NMFS, pers. comm., 2013). However, several smaller nesting sites in this region have not been sampled (e.g., Solomon Islands, Republic of Vanuatu (Vanuatu), Tuvalu, and Independent State of Papua New Guinea (PNG), etc). Within this region there is significant population substructuring (pairwise $F_{ST}$ 0.09-0.79, $p<0.05$). Of 10 nesting sites studied, four regional genetic stocks have been identified in the southwest Pacific: northern GBR, southern GBR, Coral Sea (Dethmers et al., 2006; Jensen, 2010) and New Caledonia (Dethmers et al., 2006; P. Dutton, NMFS, pers. comm., 2013; Figure 4.4). The population structure is complex, with some connectivity between northern GBR and New Caledonia ($F_{ST}$=0.117) as well as between southern GBR and the Coral Sea ($F_{ST}$=0.062); however, high genetic separation exists between these two groupings ($F_{ST}$ 0.415-0.567; Figure 4.6). Overall, this region is characterized by high nucleotide diversity.
resulting from the presence of several highly divergent lineages at these nesting sites, some of which are among the oldest lineages found in *C. mydas* (Figure 4.1).

**Figure 4.6.** Haplotype (mtDNA) frequencies (colors) at sampled nesting sites in the western Pacific (M. Jensen, NRC, pers. comm., 2013).

Traditional capture-mark-recapture studies (Limpus, 2009) and genetic mixed-stock analysis (Jensen, 2010) show that these stocks overlap on feeding grounds along the east coast of Australia. This mixing in foraging areas might provide for opportunistic mating between turtles from different stocks as evidenced by the lack of differentiation found between the northern and southern Great Barrier Reef nesting sites for nuclear DNA (FitzSimmons *et al*., 1997). Interestingly, when comparing the GBR populations to neighboring nesting sites in the Gulf of Carpentaria, both nDNA as well as mtDNA showed marked differentiation highlighting the strong reproductive isolation between the western Pacific and Indian Ocean nesting sites (FitzSimmons *et al*., 1997).

Genetic sampling in the west central Pacific has recently improved, but remains challenging, given the large number of small island and atoll nesting sites. Stock structure analysis indicates that nesting sites separated by more than 1,000 km were significantly differentiated from each other (F<sub>ST</sub> values from 0.06-0.9, p<0.001) while neighboring nesting sites within 500 km showed
no genetic differentiation. At least five management units have been identified in the region (Palau, PNG, Yap, CNMI/Guam, and Marshall Islands; Dethmers et al., 2006; P. Dutton, NMFS, pers. comm., 2013; M. Jensen, NRC, pers. comm., 2013; Figure 4.7). Nesting sites in West Central Pacific show very limited connectivity with surrounding regions (Figure 4.6).

Figure 4.7. Phylogenetic Genetic groupings (Neighbor-Joing tree of FST Values) among green turtle nesting sites in the western Pacific. Western Pacific includes the western central Pacific (WCP), central South Pacific (CSP), and southwestern Pacific (SWP). Relationships based on 384 bp of control region (M. Jensen, NRC, pers. comm., 2013).

Genetic sampling in the central South Pacific has been limited and many of the small isolated nesting sites that characterize this region have not been covered. Based on limited sampling, there is evidence of significant spatial structuring between American Samoa and French Polynesia. The samples from American Samoa were collected across four locations (Swains Island, Tutuila, Ofu, and Rose Atoll) that had both low sample sizes (n=1-8) and were a great distance from each other (160˗500 km). However, these were pooled to represent American Samoa because they shared haplotypes and were significantly distinct from French Polynesia, which was represented by one sampled nesting site (n=9) at Mopelia (P. Dutton, NMFS, pers. comm., 2013). Nesting sites from this DPS share haplotypes with the surrounding nesting sites but at low frequency. American Samoa, for example, shares haplotype CmP22.1 (59 percent), which is also found at low frequency at the Marshall Islands (17 percent) and Yap (<1 percent). Haplotype CmP65.1, which is found in both American Samoa (24 percent) and French Polynesia (99 percent), is also found in the Marshall Islands at very low frequency (<1 percent) (P. Dutton, NMFS, pers. comm., 2013). Finally, CmP47.1 was found in one individual in American Samoa (6 percent) and is found in low to high frequency at the southern GBR, northern GBR, Coral Sea and New Caledonia (Figure 4.6). There are limited data on mixed-stock foraging areas from this region. Overall, this is a region that is strongly undersampled in terms of genetics samples from both nesting sites and foraging grounds.
The key known nesting aggregations within the Hawaiian Archipelago have all been sampled. mtDNA studies show no significant differentiation (based on haplotype frequency) between French Frigate Shoals (FFS) and Laysan Island (P. Dutton, NMFS, pers. comm., 2013). While the Hawaiian Islands do share haplotypes with Revillagigedos Islands (CmP1.1 and CmP3.1) at low frequency they remain highly differentiated ($F_{ST}=0.44$) and there is little evidence of significant ongoing gene flow. The Frey et al. (2013) analysis of low level of scattered nesting on main Hawaiian Islands (MHI; Molokai, Maui, Oahu, Lanai, and Kauai; mtDNA and nDNA) showed that nesting in the MHI might be attributed to a relatively small number of females that appear to be related to each other and demographically isolated from FFS. Frey et al. (2013) suggest that the nesting population at the MHI may be the result of a few recent founders that originated from the FFS breeding population, possibly facilitated by the release of captively-reared hatching turtles into the wild.

Dispersal of animals foraging at the MHI seems also to be restricted to turtles originating from Hawaiian nesting sites with very rare records of animals from outside the central North Pacific (Dutton et al., 2008). Conversely, there is a general absence of animals from the Hawaiian breeding population at foraging areas outside the CNP (e.g., none present at Palmyra; E. Naro-Maciel, American Museum of Natural History, pers. comm., 2013).

Genetic sampling in the eastern Pacific has been extensive and the coverage in this region is substantial considering the relatively small population sizes of most eastern Pacific nesting sites. Nesting locations include mainland sites (Colola beach, Michoacán) and oceanic islands in the United Mexican States (Mexico; Socorro and Clarion Islands, Revillagigedos), the Republic of Costa Rica (Costa Rica; Nombre de Jesus) and the Galápagos Islands (Las Bachas and Las Salinas).

Among seven nesting beaches in the eastern Pacific, Dutton (NMFS, pers. comm., 2013) identified four genetic stocks based on significant $F_{ST}$ values. Their results suggest that the eastern Pacific was colonized from the western Pacific via Hawaii, and from there through the Revillagigedo Islands to mainland central America and south to the Galápagos Islands.
Figure 4.8. Haplotype (mtDNA) frequencies (colors) at sampled nesting sites in the Pacific (P. Dutton, NMFS, pers. comm., 2013; M. Jensen, NRC, pers. comm., 2013).
Recent efforts to determine the nesting stock origins of green turtles assembled in foraging areas have found that green turtles from several eastern Pacific nesting stocks commonly mix at feeding areas in the Gulf of California (Nichols, 2003; P. Dutton, NMFS, unpubl. data). Along the Pacific coast and in San Diego Bay (USA) the existing haplotype frequencies of foraging turtles suggest that these sites have substantially greater input from the Revillagigedos Islands than from Michoacán, with perhaps 100 percent of turtles coming from the Revillagigedos stock at some sites (Nichols, 2003a; P. Dutton et al., NMFS, unpubl. data). In addition, green turtles of eastern Pacific origin have been found, albeit very rarely, in Hawaiian (LeRoux et al., 2003; Dutton et al., 2008) and Japanese waters (Kuroyanagi et al., 1999; Hamabata et al., 2009). A recent study of juvenile green turtles foraging at Gorgona Island in the Republic of Colombia (Columbia) showed that most (>80 percent) of the turtles originated from nesting sites in the Galapagos islands. They also found a small contribution from Michoacán, Mexico (Amorocho et al., 2012). They also found a small number (5 percent) of turtles with the haplotype CmP22. This haplotype has never been found in any nesting population in the central or eastern Pacific, but was recently discovered to be common in nesting green turtles from the Marshall Islands and American Samoa (P. Dutton, NMFS, pers. comm., 2013). This shows that despite an apparent complete isolation of nesting females between the eastern and western Pacific nesting sites, a small number of immature turtles successfully cross the Pacific during developmental migrations. The same is true for the reverse direction: immature turtles of eastern Pacific origin have been found foraging in Japan (Hamabata et al., 2009) and New Zealand (Godoy et al., 2012).
However, it is important to point out that there is no evidence of mature turtles inhabiting foraging or nesting habitat across the Pacific.

Recent nDNA studies provide new insights that are now consistent with patterns of differentiation found with mtDNA. Roden et al. (2013) used single nucleotide polymorphism genotyping assays (n=29) developed and optimized by Roden et al. (2009) and microsatellites (n=10) characterized by Dutton and Frey (2009) to test for population structure among five Pacific green turtle nesting populations throughout the Pacific Ocean. They found significant differentiation between FFS and eastern Pacific populations (Galapagos and Michoacán) and greater connectivity between Galapagos and Michoacán than between FFS and either eastern Pacific populations. The existence of male-mediated gene flow has been assumed for green turtle populations in past studies as an explanation for low genetic structuring found with nuclear markers (Roberts et al., 2004). The results of Roden et al. (2013) contrast with those of previous nDNA studies that did not find significant population subdivision among maternally distinct populations within the Pacific based on four microsatellite loci (Roberts et al., 2004). The structure detected by Roden et al. (2013) is likely a result of increased statistical power of the tests due to the use of higher numbers of markers and larger sample sizes than the previous studies, and corroborate mtDNA studies indicating marked distinction separating FFS from the eastern Pacific populations in Galapagos and Mexico (Figure 4.7).

4.1.1.2. Tagging and telemetry

Tagging (flipper and PITs) along with satellite and acoustic telemetry provides information on movement and habitat use at various spatial and temporal scales. It is important to note that tagging studies of turtles can provide valuable information about movement of individuals, but generally they do not indicate whether these movements lead to interbreeding or gene flow. Long-term studies have primarily involved tagging females on nesting beaches. Recapture of these individuals provide, amongst other things, information on geographic range of breeding populations. Similarly, recapture of juveniles and adults tagged at foraging areas provide additional information on movement and connectivity among different habitats and regions for various life history stages. In general regional patterns emerged within ocean basins that helped inform DPS considerations. The following summarizes tagging and telemetry information for the different regions.

4.1.1.2.1. Atlantic

North Atlantic green turtle populations have minimal mixing with populations in the South Atlantic and Mediterranean regions. Occasionally juvenile turtles from the North Atlantic may settle into foraging grounds in the South Atlantic or Mediterranean. Nesters from nesting sites in the equatorial region may reside in foraging grounds in the South Atlantic (Troeng et al., 2005). Green turtles in the Mediterranean are spatially separated from populations in the Atlantic and Indian Oceans.
4.1.1.2.1.1. North Atlantic

Tagging

Long-term tagging projects exist in Bermuda (Meylan and Meylan, 2011), Costa Rica (Troëng et al., 2005), Cuba (Moncada et al., 2006), Florida (Johnson and Ehrhart, 1995; Kubis et al., 2009), Mexico (Zurita et al., 1994, 2003), the Republic of Panama (Panama; Meylan and Meylan, 2011), Puerto Rico (Patrício et al., 2011), and Texas (Shaver, 1994; 2002), and have provided a wealth of information on spatial structure for this region. Tag recovery data indicate that nesters primarily reside within the North Atlantic region; however, some nesters from equatorial beaches reside in foraging grounds in the South Atlantic (Troëng et al., 2005). There is some degree of mixing of immature turtles on foraging pastures between the North and South Atlantic.

Nesters tagged at Tortuguero, Costa Rica have been recovered throughout the Caribbean, as far north as Florida, and as far south as the Federative Republic of Brazil (Brazil). The greatest number of Tortuguero tag recoveries is from Nicaragua (Troëng et al., 2005). Turtles tagged in Cuba have also mainly been recovered in Nicaragua, and turtles that have been recovered in Cuba (i.e., tagged outside of Cuba) have predominantly been from Bermuda, the Bahamas, and “head-started” turtles from Grand Caymans (Moncada et al., 2006). There have been few recoveries of nesting females tagged in Florida at locations outside of Florida. Two Florida nesters have been recovered in Cuba and one was observed nesting in Georgia (D. Bagley, University of Central Florida, pers. comm., 2013).

Immature green turtles tagged on foraging grounds in eastern Florida have been recovered in Cuba, Nicaragua, Dominican Republic, Brazil, and Florida (D. Bagley, University of Central Florida, pers. comm., 2013). As of January 2006, 88 immature green turtles caught and tagged in Bermuda were recovered overseas in the United States, Cuba, Mexico, Venezuela, Colombia, Nicaragua, Panama, the Dominican Republic, Saint Lucia, and Grenada (Meylan and Meylan, 2011). Immature green turtles tagged in the Bahamas have been recovered in the Bahamas, Colombia, Costa Rica, Cuba, Dominican Republic, Haiti, Honduras, Nicaragua, Panama, and Venezuela (Bjorndal et al., 2003).

Telemetry

In general and based on available satellite tracking maps, green turtles that nest at North Atlantic nesting sites tend to remain primarily in this region outside of the nesting season. Nesting females from Tortuguero, Costa Rica have been tracked to Nicaragua, the Republic of Honduras (Honduras), Belize, and Mexico (Troeng et al., 2005; Sea Turtle Conservancy, 2013). Nesting turtles tracked from Lechuguillas, Veracruz, Mexico (Tiburcio Pintos et al., 2004, 2007) and Quintana Roo, Mexico (Garduño-Andrade et al., 2000; Sea Turtle Conservancy, 2013) migrated to foraging grounds off southwest Florida or remained in Mexico. Nesting females from Florida have been tracked to the Florida Keys, southwest Florida (off Cape Sable), and the Bahamas (Schroeder et al., 2008). Turtles tagged in the Cayman Islands were tracked to Belize, Mexico (Yucatan Peninsula), Honduras, Guatemala, and the Dry Tortugas National Park, FL (Blumenthal et al., 2006).
Large subadult green turtles (> 70 cm SCL) have been tracked from the east central coast of Florida to the Florida Keys, Puerto Rico, Cuba, and Bahamas (Bagley et al., 2008). A large subadult green turtle tagged in Bermuda was tracked to Cuba (Meylan and Meylan, 2011).

4.1.1.2.1. South Atlantic

Tagging

Movement between feeding grounds and nesting sites in the Caribbean and Brazil has been established by flipper tag recoveries (Lima et al., 2003, 2008).

Telemetry

In general, nesters from the eastern South Atlantic (i.e., west coast of Africa) are confined to the eastern South Atlantic and likewise for animals on the west side of the South Atlantic.

In the eastern South Atlantic, juvenile green turtles have been tracked from Corisco Bay which spans waters of the Equatorial Guinea and the Gabonese Republic (Gabon). All tracked turtles remained in the general vicinity of their release location. In Guinea Bissau, studies on reproductive behavior and satellite tracking of nesting green turtles were carried out in collaboration with the Marine Turtle Research Group, University of Wales Swansea. Nesters from Ascension Island were tracked to foraging grounds along the coast of Brazil, Oriental Republic of Uruguay (Uruguay), and the Argentine Republic (Argentina).

In the western South Atlantic, juvenile green turtles were tracked from Argentina to Uruguay and Brazil; and from Uruguay to Brazil. Four nesters from the Guianas were tracked to Brazil.

4.1.1.2.1.3. Mediterranean Sea

Tagging

Green turtles in the Mediterranean are spatially separated from populations in the Atlantic and Indian Oceans. However, few data on green turtle movements within and outside the Mediterranean are available from flipper tagging and satellite telemetry. Flipper-tagging efforts have been ongoing for over two decades at primary nesting sites in Cyprus (Demetropoulos and Hadjichristophorou, 1995; 2008) and Turkey (Y. Kaska, Pamukkale University, personal communication, 2013); however, no long-distance tagging data were encountered for use in this assessment.

Telemetry

Satellite tracking efforts have been considerably limited relative to other regions. Most satellite tracking in the Mediterranean has been on nesting females in the eastern basin. Nesters from Cyprus, Turkey, the Syrian Arab Republic (Syria), and the State of Israel (Israel) have been tracked to the Arab Republic of Egypt (Egypt), Lybia, and Turkey - with movements largely
restricted to the eastern Mediterranean (Godley et al., 2002; Broderick et al., 2007). No apparent segregation among tracked females was observed. Post-nesting females migrate primarily along the coast from their nesting beach to their foraging and overwintering grounds (Godley et al., 2002; Broderick et al., 2007).

4.1.1.2.1.4. North Indian Ocean

Tagging

Tagging of turtles on nesting beaches started in the late 1970s at Ras Al Hadd and Masirah Island, and in 1999 on the Dimaniyat Islands in the Persian Gulf. Long-term tagging and recapture records maintained on green turtles in Oman, under the Ministry of Regional Municipalities and Environment/Nature Conservation has provided information on long term movements (Ross and Barwani, 1982; Ross, 1987; Salm, 1991). No tagging has been carried out on feeding grounds (Al Saady et al., 2005). A green turtle tagged in Oman was found in the Maldives (Al Saady et al., 2005).

Telemetry

A few green turtles have been fitted with satellite transmitters within the northern Indian Ocean and reported at SEATURTLE.ORG but no data have been published. One rehabilitated female green turtle has been tracked from United Arab Emirates to east of 90° E. Because of the severe injuries to the head when it was discovered on a beach, the observed movements may not be representative. Another telemetered female green turtle remained in the coastal areas of the Persian Gulf for 49 days (N. Pilcher, Marine Research Foundation, pers. comm., 2013). Rees et al. (2012) attached satellite transmitters on two nesting female green turtles at Masirah Island, Oman. These turtles moved southward along the Arabian Peninsula and were found in Red Sea when the transmissions ceased. Telemetry data are also available for head-started green turtles at Republic of Maldives (Vabbinfaru Island, Male Atoll). These turtles have indicated wide movement patterns within the Indian Ocean (N. Pilcher, Marine Research Foundation, pers. comm., 2013).

4.1.1.2.1.5. Southwest Indian Ocean

Tagging

Evidence from tag returns indicate that while some green turtles in Tanzania are probably resident, and others are highly migratory moving to and from nesting and feeding grounds within the southwest Indian Ocean in Kenya, Seychelles, Comoros, Mayotte, Europa Island and South Africa (Muir, 2005).

Telemetry

Satellite transmitters have been deployed on green turtles at nesting beaches in the southwest Indian Ocean. Bourjea (2012) reported that green turtles nesting along the east African coast
confine their post-nesting migration to along the coast, whereas those nesting on islands (e.g., Comoros, Eparses, and Seychelles) reach the east African or Malagasy coast via “migration corridors.” This movement is believed to be mainly attributable to a network of large seagrass beds in the area. Telemetry data can be found at the following website (http://wwz.ifremer.fr/lareunion_eng/Live-Sea-Turtles).

From 2009 to 2011, 90 satellite transmitters were deployed on nesting green turtle females at five nesting sites in the southwest Indian Ocean (Europa, Glorieuses, Tromelin, Mayotte, and Moheli). Twenty percent of the tracked turtles used Madagascar coastal foraging ground while more than 80 percent used the east African coasts. The waters off north Mozambique and south Tanzania were the most important foraging ground for the tracked turtles (45 percent of the tracked turtles). Other foraging grounds included south of Maputo (Mozambique, Tulear lagoon in Madagascar; Bourjea et al., 2013).

4.1.1.2.1.6. East Indian-West Pacific Ocean

Tagging

Tagged green turtles observed in eastern Australia have been also been located elsewhere in Australia (Northern Territory, Queensland, and New South Wales) and at other neighboring countries, including Papua New Guinea, Indonesia (Java and the Anu Islands), Vanuatu, Solomon Islands, New Caledonia, and Fiji (Limpus et al., 1992, 2003, 2009; Limpus, 1993; Moritz et al., 2002; Trevor, 2009).

Telemetry

A satellite-tracked female green turtle at Redang, Malaysia, was observed to travel across areas of the open ocean, ending up near Koh Samui, Thailand (Dermawan, 2002). Other tracking studies define the range of inter-nesting habitats and post nesting migrations. Green turtles that were satellite tracked from Pulau Redang, Terengganu indicate migrations to the South China Sea and Sulu Sea areas (Dermawan, 2002).

Cheng (2000) reported movements of eight post-nesting green turtles from Wan-An Island, Taiwan using satellite transmitters. The turtles dispersed widely on the continental shelf to the east of mainland China. Destinations included southern Japan (Kyushu and Okinawa), Taiwan, and mainland China. Satellite telemetry studies demonstrated that the green turtles nesting at Taipin Tao move and forage within the southern South China Sea. Green turtle females tracked in the same area travelled long distances commencing a post-nesting migration. Eleven green turtles tracked with satellite transmitters migrated in two general directions: the first route stretched eastward along the eastern coast of the Gulf of Thailand to the Vietnam peninsula, then some crossed the South China Sea and entered Sulu Sea of Philippines water; the second route went south across the Gulf of Thailand to the Malaysian peninsula, travelling distance ranging from 456 to 2,823 km (Charuchinda et al., 2003). Finally, one study recorded post nesting migration from the China Sea north to the coastal region of Japan (Wang, 2006).
4.1.1.2.2. Pacific Ocean

4.1.1.2.2.1. Central Pacific Ocean

The Hawaiian Archipelago, including Johnston Atoll, is inhabited by green turtles that are geographically discrete in their normal range and movements, as evidenced by mark-recapture studies using flipper tags, microchip tags, and satellite telemetry. From 1965-2013, 17,536 green turtles have been tagged involving all post-pelagic size classes from juveniles to adults. With only three exceptions, the 7,360 recaptures of these tagged turtles have been within the Hawaiian Archipelago. The three outliers involved a recovery in Japan, the Marshall Islands and the Philippines (G. Balazs, NMFS, pers. comm., 2013).

Information from tagging at FFS, areas in the MHI, the NWHI to the northwest of FFS, and at Johnston Atoll show that reproductive females and males periodically migrate to FFS for seasonal breeding from these distant locations. At the end of the season they return to their respective foraging areas. FFS therefore represents the prominent focal point of green turtle nesting and hatching production in the Hawaiian Archipelago. Satellite tracking of the reproductive migrations of 19 green turtles (16 females and 3 males) illustrate the prominence of FFS to the CNP Region. All involved movements from or to FFS and the MHI. Conventional tagging using microchips and metal flipper tags has resulted in the documentation of 164 turtles making reproductive movements from or to FFS and foraging pastures in the MHI, and 58 turtles from or to FFS and the foraging pastures in the NWHI (G. Balazs, NMFS, unpubl. data).

4.1.1.2.2.2. Eastern Pacific Ocean

In the eastern Pacific (EP) flipper tagging and satellite telemetry data show that green turtle dispersal and reproductive migratory movements are generally confined to the eastern Pacific region. Long-term flipper tagging programs at Michoacán Mexico (Alvarado-Díaz and Figueroa, 1992) and in the Galapagos Islands, Ecuador (Green, 1984; P. Zarate, University of Florida, pers. comm., 2012) produced 94 tag returns from foraging areas throughout the eastern Pacific (e.g., Seminoff et al., 2002). There were two apparent groupings, with tags attached to turtles nesting in the Galapagos largely recovered along the shores from Costa Rica to Chile, in the southeastern Pacific; long-distance tag returns for the Michoacán nesting population were primarily from foraging areas in Mexico to Nicaragua. However, there was a small degree of overlap between these two regions, as at least one Michoacán tag was recovered as far south as Colombia (Alvarado-Díaz and Figueroa, 1992).

Satellite telemetry efforts with green turtles in the region have shown similar results to those for flipper tags recoveries. A total of 23 long-distance satellite tracks were considered for this assessment (Seminoff, 2000; Nichols, 2003; Seminoff et al., 2008). Satellite data show that turtles tracked in northeastern Mexico (Nichols, 2003; J. Nichols, California Academy of Sciences, unpubl. data) and California, USA (P. Dutton, NMFS, pers. comm., 2010) all stayed within the region, whereas all turtles tracked from nesting beaches in the Galapagos Islands all remained in waters off Central America and the broader southeastern Pacific Ocean (Seminoff et al., 2008).
4.1.1.3. **Morphology**

There is considerable variation in the mean nesting size (MNS) of green turtles among the numerous nesting sites worldwide (Hirth, 1997). While MNS of most populations is in the 95 cm to 110 cm CCL size range, there are a few populations that have substantially larger and smaller MNS ranges. Among populations with sufficient sample sizes, the largest turtles are found in the South Atlantic, where the MNS for green turtles at Atol das Rocas, Brazil is 118.6 cm CCL (n=738). The smallest turtles are found in the eastern Pacific, where MNS is 82.0 cm CCL in Michoacán, Mexico (n=718, (Alvarado-Díaz and Figueroa, 1992) and 86.7 cm CCL in the Galapagos (n=2708; (Zárate et al., 2003). The next smallest green turtles are found in the Mediterranean, where MNS in Alagadi, Cyprus is 92.0 cm CCL (Broderick et al., 2003).

Reported sizes of nesting females for the southwest Indian Ocean include 108 cm (median CCL) at Grande Saziley, Mayotte (Bourjea. et al., 2007), 106.3 cm (mean CCL +/- 6.1, n=61) at northern Mozambique (Garnier et al., 2007), and 108.1 cm (mean CCL +/- 5.29, n=742) at Moheli, Comoro Islands (Innocenzi et al., 2010).

Nesting turtles at Sukamade, Indonesia were measured at 99.7cm CCL (Hirth, 1997); 103.6 cm CCL at Ewu, Indonesia, 101.1cm CCL at Ashmore Reef, Australia; 98.5cm CCL at Selingaan Island, Malaysia; 99.48 cm CCL at Philippines Turtle Island; (Trono, 1991); and 82.1cm CCL at Khram Island, Thailand (Charuchinda and Monanunsap, 1998).

Kamezaki and Matsui (1995) found differences in skull morphology among green turtle populations on a broad global scale when analyzing specimens representing west and east Pacific (Japan and Galapagos), Indian Ocean (Comoros and Seychelles), and Caribbean (Costa Rica and Guyana) populations. The eastern Pacific was distinct from others based on discriminant function analysis.

Green turtles in the Hawaii population, as well as Australia, have a well-developed “crop” in the esophagus that has not been found in Caribbean or eastern Pacific populations of green turtles (Balazs et al., 1998; J. Seminoff, NMFS, unpubl. data). In addition, juvenile green turtles in Hawaii have proportionally larger rear flippers than those in the western Caribbean (Wyneken and Balazs, 1996; Balazs et al., 1998). These anatomical differences are believed to reflect adaptive variation to different environmental features in these regions as described by Balazs et al. (1998).

4.1.1.4. **Oceanographic and ecological features**

Oceanographic and ecological features of turtle habitats can be relevant for considerations of both discreteness and significance. We provide a comprehensive summary of these features in this section; in subsequent sections, we highlight features that are particularly informative regarding discreteness or significance.
Mediterranean Sea

The Mediterranean Sea is a remnant of the ancient Tethys seaway that provided a tropical connection between the Atlantic and Indo-Pacific Oceans. Northward drift of Africa to join Eurasia around 10 mya closed the Indo-Pacific link, and about 6 mya communication with the Atlantic was also closed, resulting in an isolated inland sea that went through several extreme evaporative cycles. During this period, which is termed the Messinian Salinity Crisis (Ruggieri et al., 1967), most of the Indo-Pacific biota was extirpated, so current biota is derived primarily from re-invasion of Atlantic species following re-opening of the narrow connection through the Strait of Gibraltar about 5 mya. Today, the Mediterranean Sea is a virtually enclosed basin with warm (average temperatures 15-21°C) and salty (average salinity 36.2-39 ppt) water. It is rich in oxygen but poor in nutrients, and this pattern of oligotrophy increases from west to east. The Mediterranean is typically split into three basins: the western Mediterranean, the Adriatic, and the eastern Mediterranean. Marine community structure in the Mediterranean differs considerably from that in the Atlantic, and Mediterranean populations often have smaller-sized individuals (Hirth, 1997). Because of its exceptionally high levels of endemism and critical levels of habitat loss, the Mediterranean Sea is one of 25 biodiversity centers recognized on a global scale (Myers et al., 2000); it is also widely recognized as an area among the most sensitive to climate change (UNEP-MAP RAC/SPA, 2010).

Atlantic Ocean

In the Atlantic Ocean, green turtles nest on continental and island beaches between about 30° N and 30° S latitudes (Witherington et al., 2006). Juvenile turtles can be found as far north as Cape Cod Bay, Massachusetts, in Bermuda to the east, and throughout the Caribbean. Water temperatures below 8°C result in hypothermic cold-stunning (Witherington and Ehrhart, 1989) and thus serve as a natural seasonal boundary. Green turtles take advantage of the warm waters of the Gulf Stream to nest in North Carolina at 34°N, which is farther from the equator than any other nesting sites outside the Mediterranean Sea.

Green turtle foraging grounds in the North Atlantic Ocean range from coral or nearshore reefs and seagrass beds, to inshore bays and estuaries, to man-made embayments (Guseman and Ehrhardt, 1990; Bresette et al., 2002; Ehrhart et al., 2007; Kubis et al., 2009). Turtles feed primarily on sea grass or benthic macroalgae depending on the habitat in which they reside (Bjorndal, 1980; Mortimer, 1981; Coyne, 1994; Shaver, 1994; Redfoot, 1997). The quality of nesting beach habitat ranges from undeveloped, natural coastlines to developed and armored shores. In Florida, green turtles seem to prefer barrier island beaches that receive high wave energy and have coarse sands, steep slopes, and prominent foredunes (Witherington et al., 2006).

In the southern hemisphere nesting occurs on beaches in South America and on Caribbean islands in the western Atlantic, and along the coast of Africa in the eastern Atlantic. Nesting also occurs on an oceanic island (Ascension Island) on the mid-Atlantic ridge, and turtles forage on coastal sea grass beds in Brazil. The cold Benguela Current provides a barrier to reproductive movement between the southeast Atlantic and southwest Indian Ocean, with dispersal believed to only occur over evolutionary timescales (see below).
Indian Ocean

Water enters the Indian Ocean from the Atlantic and, through the Indonesian seas, from the Pacific. The principal upper ocean flow is dominated by two regimes: (1) the subtropical anticyclonic gyre of the southern Indian Ocean, and (2) monsoonally-forced circulation of the tropics north of the equator. These two regimes are separated oceanographically at approximately 10-12° S by a nearly zonal current (Southern Equatorial Current) carrying fresher Pacific waters westward across the Indian Ocean. The western boundary current (Agulhas Current) overshoots the African coast. The eastern boundary current (Leeuwin Current) flows toward the south. The surface waters of the tropical Indian Ocean are the warmest of the global open oceans, often exceeding 29°C. Water temperature in the upper layer is highest between 20° N and 20° S, except along the western boundary, where upwelling occurs north of the equator along the Horn of Africa. South of the equator, relatively high temperature extends southward along the western boundary (Reid, 2003). Salinity is highest west of India due to excess evaporation, whereas the lowest salinities are found in the area of high rainfall north of the equator along the eastern boundary of the basin (Reid, 2003). Oxygen is close to 4.6 ml/l north of 25°S and rises to more than 8 ml/l in the colder water near Antarctica (Reid, 2003).

In the tropics and northwest Indian Ocean, the circulation is strongly seasonal, forced by the reversing of southwest and northeast monsoons. The Arabian Sea is saline and its marginal seas (Red Sea and Persian Gulf) are dominated by evaporation. Within the Arabian Sea, circulations are cyclonic in December-February.

The main oceanographic feature in the southwest Indian Ocean is upwelling along the western boundary. Oceanographic conditions at the northern and southern parts of the Mozambique Channel are different enough that dispersal of turtle hatchlings may be affected (Bourjea et al., 2007). Hatchlings emerging from nests on the southern and western side of the Mozambique Channel should drift southward following large anticyclonic eddies (Bourjea et al., 2007). On the eastern side, however, the flow is weak and variable. In the northern part of the Channel, the flow is variable but on average forms an anticlockwise gyre in the Comoro Basin that becomes part of the East African Coast Current as it flows northward along the continent (Gordon et al., 1997; Bourjea, 2007). Currents around Europa, Eparses Islands, act as a barrier for adult green turtles (Girard et al., 2006). This region is also characterized by contrasts between areas of upwelling and areas of increased sea-surface temperatures, the occurrence of meanders, and a convergence zone between different currents. Collectively, these oceanographic features probably contribute to the genetic structure of green turtles in and around the Mozambique Channel (Bourjea et al., 2007).

In the east, the Bay of Bengal is fresher than the Arabian Sea because of the runoff from all of the major rivers of India, Bangladesh, and Burma. Within the Bay of Bengal, circulations are cyclonic in December-February.

Because the Indian subcontinent extends only to about 8° N latitude, vast areas of the central and southern Indian Ocean do not have suitable benthic foraging habitat for adult green turtles.
Oceanography of the world’s largest ocean is dominated by two large gyres extending from near the equator halfway to the poles. The gyres, which circulate clockwise in the northern hemisphere and counterclockwise in the southern hemisphere, have high pressure areas at the center, which produce winds that drive surface currents that flow from east west on either side of the equator. The central parts of these gyres act as sinks for atmospheric CO₂ and also concentrate anthropogenic pollutants. The North Pacific subtropical gyre is the largest ecosystem on the planet.

Tropical marine habitats in the eastern Pacific are restricted by incursion of cold water from two strong currents flowing from the poles toward the equator: the California Current in the north, and the Humboldt Current in the south. Because the west side of the South American continent is oriented almost directly north-south, the Humboldt Current penetrates into the equatorial regions. In contrast, south of central California, USA, the North American continent bends sharply to the east, and this leaves a tropical region in Central American and northern South American waters that is less affected by upwelling (Briggs, 1974). One notable feature that distinguishes green turtle habitats in the eastern Pacific Ocean is the extremely narrow continental shelf. This limited area allows cold upwelled water to have a greater influence on coastal neritic habitats than in other regions. Areas such as the Galapagos Islands and continental shelves of the USA, Mexico, Ecuador, Peru, and Chile experience unusually cool waters relative to other areas of similar latitude. The limited shelf areas also contribute to this region’s nearly-complete lack of seagrasses, a primary habitat and diet component of green turtles in many other regions (Bjorndal, 1980).

One of the world’s best-known biogeographic barriers is the Wallace Line, which corresponds to a deep-water channel that separates New Guinea and Australia from Borneo and Southeast Asia. In most configurations, the line also continues northward and passes to the east of the Philippines (Mayr, 1944).

As noted by Briggs (1974), “the East Pacific Barrier is the formidable stretch of deep water that lies between Polynesia and America.” Biogeographic studies indicate the effectiveness of the barrier, measured in terms of co-occurrence of species on both sides, exceeds 90 percent for a variety of marine taxa (Briggs, 1974). The vast expanses of open ocean that isolate eastern Pacific populations are also generally considered to have low productivity, and this could be particularly important for green turtles, which have long been considered obligate neritic inhabitants (Hirth, 1997). However, recently it has become apparent that some green turtles are high-seas dwellers and perhaps live a significant portion of their juvenile and adult lives in waters far from shore (Seminoff et al., 2008).

4.1.1.5. Summary of conclusions regarding Discreteness

The SRT recognized that discreteness within taxonomic species can be assessed at a variety of hierarchical scales. At one extreme are population segments that have been completely or almost completely isolated for long evolutionary periods of time. At the other extreme are local breeding units that might be demographically independent on ecological time scales (in the sense
that population dynamics are driven more by local births and deaths than by immigration) but nevertheless regularly exchange some migrants and genes with other such units. Units that occur along any part of this continuum might potentially meet the discreteness criteria in the joint policy, but the probability that the unit, if considered discrete, would also be considered significant increases with the strength and duration of isolation. Accordingly, the SRT began its evaluations of discreteness by focusing on the strength of the phylogeographic signal, i.e., the degree of congruence between geography and the distribution of genetically divergent population units. The major patterns are summarized below, with more details provided in the summaries for each DPS.

Five dominant and geographically widespread clades and three geographically restricted clades were identified in the mtDNA analysis (Figure 4.1). These clades reflect divergence times measured in hundreds of thousands to millions of years. Each of the following clades is strongly associated with specific geographic regions: Mediterranean and western North Atlantic (Clade I; yellow), the rest of the Atlantic (Clade II; gray), the Indian Ocean and parts of the NW Pacific (Clade VII; blue), the East Pacific and Hawaiian Islands (Clade VIII; green), and Micronesia, and Melanesia (Clade IV; red). Three clades are rare and occur only in restricted geographic areas or are scattered geographically but occur in low frequency. These include Clade III (rose) found in Southern Polynesia, New Caledonia, the Great Barrier Reef, Japan, Coral Sea, and through Southeast Asia in low frequency; Clade V (purple) found only in nesting sites in the Coral Sea (Great Barrier Reef, Coral Sea and New Caledonia), and Clade VI (light blue) commonly found in northern and western Australia. The Persian Gulf consists of divergent haplotypes without a strong position in the tree (highlighted by black/white shading). More sampling from the northwest Indian Ocean region will provide a more robust placement of those haplotypes.

All of the major clades include a large number of different haplotypes, so populations that have haplotypes from the same clade are not necessarily genetically similar. In many cases, different regional populations within the same clade have nearly or completely non-overlapping sets of haplotypes: Mediterranean vs. western North Atlantic (Figure 4.2), and east Pacific vs. Hawaii (Figure 4.8) are examples of this. Although most green turtles from the Indian Ocean have haplotypes derived from a single clade, substantial regional differences also are found in the occurrence of specific haplotypes (Figure 4.4).

Geographic differences in mean nesting size (MNS), skull morphology and anatomy provide supporting evidence for the eastern Pacific populations being discrete from the others. However, there are limitations in sampling distribution and the efficacy of these traits as diagnostic tools to discriminate populations by themselves. Differences in MNS of Mediterranean, South Atlantic and some Indian Ocean populations also support distinction at these broader regional scales.

Tagging and telemetry studies show that North Atlantic green turtle populations have minimal mixing with populations in the South Atlantic and Mediterranean regions. Occasionally juvenile turtles from the North Atlantic may settle into foraging grounds in the South Atlantic or Mediterranean. It is extremely rare for nesters from nesting sites in the equatorial region to reside in foraging grounds in the South Atlantic.
Naturally occurring biological and physical barriers clearly play a role in structuring some green turtle populations. The apparent lack of nesting sites in the western Mediterranean and eastern Atlantic north of the equator creates a gap of many thousands of kilometers between populations in the eastern Mediterranean and those anywhere else in the world. Significant gaps in nesting sites that appear to act as isolating mechanisms also occur along the coast of southwest Africa, along the Horn of Africa, along the east coast of India, and along much of the western shore of South America. The eastern Pacific and Hawaiian populations are separated by the East Pacific Barrier, and the expanse of deep water in the southern Indian Ocean isolates populations from either side of that basin. Populations from a vast area in the South Pacific and western Pacific are generally dominated by mtDNA haplotypes from a clade that is rare or missing in populations from other areas. Population boundaries in the western Pacific are consistent with the Wallace Line (Mayr, 1976), an established transition zone separating the fauna of Asia and Australia.

Collectively, these observations led the SRT to propose that green turtles from the following geographic areas might be considered “discrete” according to criteria in the joint policy:

1. North Atlantic Ocean
2. Mediterranean Sea
3. South Atlantic Ocean
4. Southwest Indian Ocean
5. North Indian Ocean
6. East Indian Ocean-West Pacific Ocean
7. Central West Pacific Ocean
8. Southwest Pacific Ocean
9. Central South Pacific Ocean
10. Central North Pacific Ocean
11. East Pacific Ocean

A formal vote (see Table 4.1) showed that each of these population units received at least 70% affirmative votes from the SRT. Therefore, the SRT determined that each of these units could be considered discrete from conspecific population segments of *C. mydas* for purposes of the ESA. Figure 4.8 depicts these units. The next section explains how each of these population units was evaluated in terms of significance.

**Comment [A16]:** The term “isolating mechanisms” has fallen out of favor in literature on population isolation because of teleological implications; it implies that species evolve such barriers to isolate themselves.

**Comment [A17]:** Response: revised

**Comment [A18]:** I suggest that explicit information be provided of the component management units or sub-areas of each of these 11 discrete geographic areas, so as to clearly define their geographic extents. This is especially important for DPSs 7 through 9 and for component areas near the boundaries of the DPSs. Figure 4.10 is not sufficient for this purpose. Figures 4.3, 4.5, and 4.7 identify component sub-areas only for those for which genetic data are available.

**Comment [A19]:** Response: I’m not sure if this can be done succinctly or is needed. If I understand, the component areas are basically the nesting sites (populations), and these are given in map that shows all known nesting sites Fig. 4.10...but this reviewer is suggesting adding a comprehensive table/list of all the “components”, including marine-related features??? I think this was part of the broader delineation discussion of where to draw boundaries.
Table 4.1. Results of SRT voting for discreteness. Values show the mean of affirmative likelihood points (with range among SRT members in parentheses).

<table>
<thead>
<tr>
<th>DPS</th>
<th>Discreteness</th>
</tr>
</thead>
<tbody>
<tr>
<td>North Atlantic</td>
<td>83.8 (55-95)</td>
</tr>
<tr>
<td>Mediterranean</td>
<td>96.5 (90-100)</td>
</tr>
<tr>
<td>South Atlantic</td>
<td>84.1 (60-90)</td>
</tr>
<tr>
<td>Southwest Indian</td>
<td>72.9 (50-90)</td>
</tr>
<tr>
<td>North Indian</td>
<td>81.3 (65-95)</td>
</tr>
<tr>
<td>East Indian-West Pacific</td>
<td>71.2 (40-90)</td>
</tr>
<tr>
<td>Central West Pacific</td>
<td>70.9 (50-90)</td>
</tr>
<tr>
<td>Southwest Pacific</td>
<td>79.9 (40-99)</td>
</tr>
<tr>
<td>Central South Pacific</td>
<td>70.0 (50-95)</td>
</tr>
<tr>
<td>Central North Pacific</td>
<td>93.7 (85-100)</td>
</tr>
<tr>
<td>East Pacific</td>
<td>91.6 (75-100)</td>
</tr>
</tbody>
</table>

Figure 4.10. Map of all C.mydas nesting sites indicating delineation of DPSs.

4.1.2. Significance Determination

Below we consider whether the 11 discrete population segments identified in Section 4.1.1.5 can also be considered significant. Our discussion is oriented around criteria identified in the joint DPS policy.

4.1.2.1. Ecological setting

Here we briefly summarize information presented in Section 4.1.1.4 and elsewhere that is particularly relevant to evaluating significance.

Areas with a large fraction of endemic species support novel biological communities that can have a profound effect on both ecological and evolutionary processes. Tropical marine areas that
support green turtles and that have unusually high degrees of endemism of marine species include the Mediterranean Sea, the east Pacific, and the Hawaiian Islands.

The Mediterranean Sea provides the most highly saline marine waters for green turtles, while high temperatures and evaporation rates also produce high salinities in the Red Sea and Persian Gulf. Conversely, high rainfall and extensive freshwater runoffs from major rivers produce relatively low salinity water in the northeastern Indian Ocean.

Major upwelling areas that occur in the Mozambique Channel and along the west coasts of North and South America infuse cold water into otherwise tropical areas and create distinctive habitats for green turtles. Convergence of ocean currents around Madagascar also creates complex oceanographic patterns in this area. The lack of significant continental shelf habitat in the eastern Pacific also affects sea turtle ecology in that area. Oligotrophic, low-productivity waters in the eastern Mediterranean and in the large oceanic gyres in the north and south central Pacific and in the southern Indian Ocean present challenges for turtles foraging in those areas.

In the south central Pacific, nesting habitats are spread over a particularly large geographic area. Green turtles in this area have few substantial island habitats that can serve as stronghold populations and instead are distributed across many small atolls and islands to a greater extent than occurs in any other area.

4.1.2.2. Gap in the species range

Because each of the discrete population segments identified above occupies all or a large portion of one of the major ocean basins in the world, it could be argued that loss of any of these units would represent a significant gap in the global range of green turtles. This argument would be particularly strong for the following discrete units: Mediterranean (the species would be lost from this entire ocean basin); South Atlantic (would create a gap of at least 12,000 kilometers between populations off southeast Africa and those in Florida); Eastern Pacific, Central North Pacific, and Central South Pacific (loss of any would create a large gap in the world’s largest ocean basin); and East Indian-West Pacific (loss of turtles from this large and complex area that includes what is likely the center of origin for the species would substantially reduce connectivity among remaining populations).

4.1.2.3. Marked genetic differences

Because neutral genetic markers were already used heavily in defining discreteness, the SRT felt the same data were potentially relevant for evaluating significance only if the differences were quite substantial and strongly coherent geographically. From this perspective, the strongest evidence for marked genetic differences exists for Eastern Pacific and Central North Pacific: all turtles in these areas carry haplotypes from a divergent clade not found anywhere else in the world. Strong cases can also be made that the following areas also meet this criterion, despite some slight fuzziness in the geographic boundaries of the mtDNA clades: Mediterranean, North Atlantic, and South Atlantic.
4.1.2.4. Other factors

The joint DPS policy acknowledges that other types of information beyond those identified in the above criteria can be useful for evaluating significance of discrete population units. In this section we consider aspects of behavior and life history that might suggest local adaptations.

Size of nesting turtles

Substantial differences in the size of nesting females suggests local adaptations or conditions. Nesters from nesting sites in the North Atlantic are larger (105 cm CCL) than those in the proximate Mediterranean Sea (average CCL 88-96 cm). Immediately to the south, nesters at varying locations in the South Atlantic are larger than those in the North Atlantic or Mediterranean. For example, nesters at Isla Trindade, Brazil (average CCL 116.8 cm), Atol das Rocas, Brazil (average CCL 118.6 cm), and Ascension Island (average CCL 116.8 cm) are among the largest nester sizes reported for green turtles globally (Hirth, 1997a). In the Indian Ocean, nesting sites in the southwest and northern populations also have substantial differences in nester size (see Table 16.1, also van Buskirk and Crowder, 1994).

Nesting season

The nesting season varies slightly among populations in the eastern Indian and western Pacific Oceans, with the main season being from June to August. Green turtle nesting is from March to July in the inner Gulf of Thailand, (Abe et al., 2003; Charuchinda et al., 2003; Aureggi et al., 2004). Nesting season is during the summer months in North Atlantic populations.

Clutch Frequency

The clutch frequency for Ascension Island nesters was re-evaluated in 2012; 40 females were outfitted with radio transmitters in an effort to quantify clutch frequency. Those females laid an average of 6.3 clutches per season, more than double the previous estimate.

Behavior

Because of the temperate nature of many green turtle foraging areas at the northern and southern extents of their range in the eastern Pacific, green turtles may experience colder waters in this region than anywhere else in the world. For example, in northwestern Mexico and California, USA, green turtles become inactive during the cold months of December to March (Seminoff, 2000). During this period, green turtles may enter a torpid state during which they may lay motionless on the sea floor for days to weeks. This behavior is poorly understood, although green turtle overwintering is the focus of increasing study and has also been documented in the Mediterranean and Gulf of Mexico (Broderick et al., 2007; Hochscheid et al., 2007).

A common behavioral trait that appears to characterize green turtles in the Galapagos Islands, Hawaii, and Australia is basking, where turtles haul out on beaches or sand dunes during the daytime to apparently warm in the sunlight. This behavior is rare in other parts of the world and
possibly is an adaptive response to the cooler thermal environment and ocean predation pressure in those regions (Whittow and Balazs, 1982; Green, 1998; Limpus, 2008).

Remigration interval

Female turtles in the North Atlantic have relatively short migration distances and typically have 2-year remigration intervals (Witherington et al., 2006), whereas 3-year or longer intervals are more common elsewhere. Re-migration interval has a large effect on population dynamics, population viability, and recovery potential.

Disease

The fibropapillomatosis (FP) disease appears to affect green turtles primarily in the central North Pacific Ocean (Chaloupka et al., 2009; Francke et al., 2013), and southeastern U.S (Hirama and Ehrhart, 2007), where a large fraction of individuals can be affected. This pattern may reflect genetic distinctiveness or distinctive aspects of their habitats.

Latitude

All nesting sites within the Mediterranean are at latitudes 31-40°N, which is outside the normal latitudinal range for this species. In addition to effects on temperature, latitude strongly affects variation in day length and seasonality of environmental conditions, which are likely to have fostered local adaptations in green turtles living there.

4.1.2.5. Summary of conclusions regarding Significance

Although the joint policy refers to ecological settings that are “unusual or unique” for the species, if enough variables are measured, every ecological setting can be considered unique in the sense of “one-of-a-kind.” Therefore, the SRT focused on evaluating the degree to which each discrete population segment occupies habitats with unusual or distinctive ecological features. As noted in Sections 4.1.2.1, discrete population units with particularly distinctive ecological features include the Mediterranean, North Indian, Southwest Indian, East Pacific, Central North Pacific, and Central South Pacific.

Loss of populations from the following areas would leave particularly large gaps in the global distribution of green turtles: Mediterranean, South Atlantic, East Pacific, Central North Pacific, Central South Pacific, and East Indian-West Pacific. Distinctive features of morphology, behavior, or life history that might indicate important local adaptations are documented for turtles from the Mediterranean, North Atlantic, South Atlantic, North Indian, East Pacific, and Central North Pacific. The strongest case for occurrence of marked genetic differences can be made for green turtles from the following areas: East Pacific, Central North Pacific, Mediterranean, North Atlantic, and South Atlantic. Nesting turtles are smaller in the Mediterranean and North Indian Ocean, while those from many sites within the North and South Atlantic are larger. Turtles from the East Pacific are morphologically distinctive and exhibit a unique behavior somewhat akin to hibernation. All nesting sites in the Mediterranean and some in the North Atlantic are outside the normal latitudinal range for the species.
After considering all of the above information, the SRT voted on significance, and each discrete population unit identified earlier received a substantial majority (65% or more) affirmative votes (Table 4.2). Therefore, the SRT concluded that each of the 11 discrete population units is also significant to the species to which it belongs, *C. mydas*, and therefore satisfies the criteria for being a DPS.

**Table 4.1.** Results of SRT voting for significance. Values show the mean of affirmative likelihood points (with range among SRT members in parentheses).

<table>
<thead>
<tr>
<th>DPS</th>
<th>Significance</th>
</tr>
</thead>
<tbody>
<tr>
<td>1 North Atlantic</td>
<td>82.1 (70-100)</td>
</tr>
<tr>
<td>2 Mediterranean</td>
<td>96.5 (85-100)</td>
</tr>
<tr>
<td>3 South Atlantic</td>
<td>85.8 (65-100)</td>
</tr>
<tr>
<td>4 Southwest Indian</td>
<td>65.4 (40-95)</td>
</tr>
<tr>
<td>5 North Indian</td>
<td>81.3 (65-95)</td>
</tr>
<tr>
<td>6 East Indian-West Pacific</td>
<td>86.7 (60-99)</td>
</tr>
<tr>
<td>7 Central West Pacific</td>
<td>70.0 (50-95)</td>
</tr>
<tr>
<td>8 Southwest Pacific</td>
<td>83.2 (75-90)</td>
</tr>
<tr>
<td>9 Central South Pacific</td>
<td>77.1 (60-95)</td>
</tr>
<tr>
<td>10 Central North Pacific</td>
<td>96.2 (85-100)</td>
</tr>
<tr>
<td>11 East Pacific</td>
<td>95.8 (90-100)</td>
</tr>
</tbody>
</table>

Although DPS is a legal term and the SRT recognizes that these population segments are not technically DPSs until or unless they are designated as such in a rulemaking process, for lack of a better term, we refer to these units as DPSs throughout the report.

4.2. **Detailed summaries of discreteness and significance determinations for each DPS**

4.2.1. **North Atlantic DPS**

*Discreteness*

Most turtles in this DPS carry haplotypes from a lineage that is found only here and in the Mediterranean. Populations in the Mediterranean are at least 8,000 km away and share only one specific haplotype (found in two individuals) with North Atlantic populations. The nearest populations outside this DPS are found in the eastern Caribbean, but these and all other populations from the South Atlantic DPS carry mostly mtDNA haplotypes from a different clade (II), indicating strong long-term isolation. Tagging studies have identified juveniles from this DPS in waters off Brazil and Argentina, but we found no evidence of movement of mature individuals.
Significance

This region is characterized by a broad continental shelf that provides abundant pastures for *Thalassia* species (“turtle grass”) that the turtles depend on for food. Turtles in this DPS can be considered to differ markedly in their genetic characteristics, given that they are strongly divergent from other populations within Clade I (from the Mediterranean; Figure 4.1), and turtles from adjacent populations in the eastern Caribbean carry haplotypes from a different clade. Nesting sites in northern Florida and North Carolina are farther from the equator than any other green turtle sites outside of the Mediterranean Sea. The re-migration interval for females from several sites within the DPS is shorter (typically 2 yrs) than that reported for other DPSs.

4.2.2. Mediterranean DPS

Discreteness

With the only outlet through the narrow Strait of Gibraltar, the Mediterranean Sea is the most isolated ocean basin in the world. Furthermore, existing populations of green turtles inhabit only the eastern portion of the sea and thus have little opportunity to interact with turtles from other areas. No turtles tagged in the eastern Mediterranean have been recovered farther west than the Tunisian Republic (Tunisia) inside the Mediterranean. The nearest populations outside the Mediterranean are several thousand kilometers away in the Republic of Senegal (Senegal). Mediterranean turtles all carry mtDNA haplotypes from Clade I, and the only other place this lineage is found is in the North Atlantic. North Atlantic and Mediterranean turtles, however, do not share any individual haplotypes (except from two individuals, one from Cuba and one from Turkey sharing the same haplotype), which indicates very strong long-term isolation of females.

Significance

The Mediterranean Sea is the saltiest ocean basin in the world and, being nearly enclosed, provides a unique ecological setting for the species. Loss of green turtles from this large area would create a major gap in the species’ range. Given the strong genetic divergence and distinctive environmental conditions, it is likely that turtles from the eastern Mediterranean have developed local adaptations that help them persist in this area. Nesting females in this DPS are smaller than in any other DPS except the Eastern Pacific.

4.2.3. South Atlantic DPS

Discreteness

With a few exceptions, every green turtle in this DPS carries an mtDNA haplotype from a clade that is found nowhere else (Clade II), indicating strong isolation of matriline over evolutionary time periods. The exceptions to this pattern are: (1) one population from the eastern Caribbean has a low frequency of a haplotype from the North Atlantic - Mediterranean Clade (I), and (2) populations from the Gulf of Mexico/Central America have a low frequency of haplotypes from the South Atlantic Clade (II); (2) two populations from southeast Africa have high frequencies of haplotypes from the South Atlantic Clade. We believe these reflect historical events rather than
contemporary connectivity. This interpretation is supported by satellite telemetry, which reveals extensive movements of turtles within the south Atlantic region but no evidence for migrations into other DPSs, other than rare instances of movement into foraging areas in the North Atlantic. Long stretches of cold water along the coasts of Patagonia and southwest Africa serve to isolate South Atlantic turtles from populations in the east Indian and east Pacific oceans.

**Significance**

The average size of nesting females is larger here than in any other DPS, which could reflect adaptation to local environmental conditions. The substantial population at Ascension Island is the one of very few nesting sites in the world associated with a mid-ocean ridge. Loss of all green turtles from this vast area would create a gap of at least 12,000 kilometers between populations off southeast Africa and those in Florida.

4.2.4. Southwest Indian DPS

**Discreteness**

Although the two southernmost populations that have been sampled contain some haplotypes from the South Atlantic Clade, this reflects ancestral relationships rather than contemporary connectivity. The expanse of cold water along the southwest coast of Africa represents a barrier with Atlantic populations. Connections to the east are inhibited by the expanse of open water in the southern Indian Ocean, and to the north an apparently uninhabited stretch along the Horn of Africa divides this DPS from the North Indian DPS. One haplotype common in the North Mozambique Channel is found throughout Southeast Asia, indicating historic connectivity. However, tagging and tracking data document movements within but not between DPSs, except for a small proportion of trans-boundary movement between this DPS and populations in the North Indian Ocean.

**Significance**

Strong upwelling in the Mozambique Channel produces distinctive areas of high productivity that support robust turtle populations, and complex current patterns in the area create a distinctive ecological setting for green turtles. Madagascar is one of the largest islands in the world and its proximity to the African coast, along with a proliferation of nearby islands, create a complex series of habitats suitable for green turtles. Nesters in this DPS are larger than in other DPSs within the Indian Ocean, which could reflect local adaptations. This DPS has a high degree of genetic diversity, with haplotypes present at nesting sites from three divergent evolutionary clades. Loss of all turtle populations from this DPS would leave a gap of over 10,000 km between populations in southern India and those in west-central Africa.
4.2.5. North Indian DPS

Discreteness

Genetic data are very limited for this DPS; the only sample is from the Persian Gulf, and it has two groups of highly divergent haplotypes that are not found anywhere else in the world. This DPS appears to be isolated from other Indian DPSs by substantial breaks in nesting habitat along the Horn of Africa and along the entire eastern side of the Indian subcontinent.

Significance

Nesting turtles here are smaller than in other Indian DPSs. This region contains some of the warmest and most saline waters in the world. This region was characterized by only a single sample (from the Persian Gulf) in our mtDNA study, but contains unique haplotypes that seem to form two additional clades. However, bootstrap support for these clades is weak due to a small regional sample. Additional samples are needed in order to assess their place in the phylogenetic tree.

4.2.6. East Indian-West Pacific DPS

Discreteness

This is a large and complex DPS that contains the core of the global distribution of green turtles (Figure 4.84). Most populations are dominated by haplotypes from Clade VII, but with some overlap of Clades III and IV throughout the Indian Ocean; evidence of a complex colonization history in this region. While one common haplotype is shared across the Indian Ocean, substantial gaps in nesting sites along the east coast of India and in the southern Indian Ocean serve to isolate this DPS from those in the north and southwest Indian Ocean. The Wallace Line and its northern extension separate this DPS from populations to the east, which carry primarily from Clade IV. Nesting sites to the northern extreme (Taiwan and Japan) show more complex patterns of higher mixing of divergent haplotypes, and the placement of individual nesting sites within this DPS is somewhat uncertain and may become better resolved when additional genetic data is available.

Significance

This area of complex habitats at the confluence of the tropical Indian and Pacific Oceans is a well-known hotspot for speciation and diversification of both terrestrial and marine taxa, and this enormous diversity creates a distinctive ecological setting for green turtles. Loss of all populations from this vast area would create a substantial gap in the global distribution and, because this DPS is so centrally located, would strongly affect connectivity within the species as a whole.

Comment [A20]: Shouldn't this be Figure 4.4?
Comment [A21]: Response: Appreciate this reviewers attention to detail in catching this!!
Comment [A22]: Placement of these sites within EIO/WP rather than within Central West Pacific needs more discussion. Placement appears to be based on spatial proximity. Judging from information in Figure 4.1 and, to a lesser extent Figure 4.4, these sites are very difficult to place. I do not think that their placement invalidates either presumptive DPS (they would be a very minor part of either), but their problematic identity needs to be acknowledged and an explicit statement of the rationale for their placement should be provided.
Comment [A23]: Response: Agreed, based on limited analysis these sites are difficult to place.
4.2.7. Central West Pacific DPS

Discreteness

Some tag recoveries indicate movement of adults between this DPS and the East Indian-West Pacific DPS. However, this DPS carries haplotypes from Clade IV, while those to the west carry mainly haplotypes from Clade VII, so these presumably reflect foraging migrations rather than interbreeding. The boundary between this DPS and the East Indian-West Pacific DPS is congruent with the northern portion of the Wallace Line. Wide expanses of open ocean separate this DPS from the Central North Pacific DPS, and genetic data provide no evidence of gene flow over evolutionary time scales. Tagging studies also have not found evidence for migration of breeding adults to or from the adjacent DPSs (see details summary in Section 16).

Significance

The geographic area included in this DPS encompasses most of the area commonly referred to as Micronesia, as well as parts of Melanesia. Like DPSs 9 (southern Polynesia or Central South Pacific) and 10 (northern Polynesia or Central North Pacific), the central West Pacific has no continental shelf habitats, so all nesting is on small islands or atolls. Loss of turtles from this DPS would create a large hole near the center of the geographic range of the species.

4.2.8. Southwest Pacific DPS

Discreteness

This DPS is characterized by haplotypes from Clade V which has only been found at nesting sites in this DPS. It also has high frequency of haplotypes from Clades III and IV, as well as low frequency of haplotypes from Clades VI and VII making this area highly diverse. Haplotypes from the widespread Clade IV also are common in DPSs 7 and 9, but consisting of different haplotypes. Tagging, telemetry, and genetic studies show movement of breeding adults occurs mainly within this DPS.

Significance

Unlike most other DPSs in the Pacific Ocean, this DPS includes a mix of island nesting sites and coastal foraging areas. The GBR, the largest coral reef system in the world, provides a unique ecological setting for nesting as well as foraging green turtles. The northern GBR supports one of the largest nesting populations in the world, but the majority (>90%) of the nesting occurs on one small island (Raine Island).
4.2.9. Central South Pacific DPS

**Discreteness**

This DPS is isolated by vast expanses of open ocean from turtle populations to the north (Hawaii) and east (Galapagos), and in both of these areas turtles are fixed for haplotypes from a different clade (Clade VIII). To the west, this DPS adjoins the Central West Pacific and Southwest Pacific DPSs. Genetic samples are available from only two nesting populations within this DPS, but they both contain relatively high frequencies of haplotypes from a single clade (Clade III) that is not found in either of the latter DPSs.

**Significance**

This area represents a substantial part of the South Pacific subtropical gyre ecosystem. To a greater extent than in any other DPS, nesting sites are widely dispersed among a large number of small habitats on tiny islands and atolls. Although turtles in this area are poorly studied, it is likely that they have evolved adaptations to persist with this very diffuse metapopulation structure. If green turtles were lost from this entire DPS, it would create a large gap in the range across the southern ocean.

4.2.10. Central North Pacific DPS

**Discreteness**

The Hawaiian archipelago is the most isolated group of islands in the world (Juvik et al., 1999). Genetic data indicate that this isolation also serves as a barrier to migration of green turtles, as mtDNA haplotypes from the Hawaiian Islands are from an evolutionarily divergent clade that is restricted to Hawaii and the Eastern Pacific. Extensive tagging data have not shown movements between Hawaii and other locations in the Pacific. The East Pacific Barrier, which greatly restricts or eliminates gene flow for most marine species from a wide range of taxa (Briggs, 1974), also appears to act as a barrier to movement of green turtles. Only a tiny fraction of mtDNA haplotypes is shared between Hawaiian and east Pacific populations, likely reflecting connectivity over deeper evolutionary timescales (>1 million year). Nuclear data also show a marked separation between Hawaii and the eastern Pacific nesting populations. Turtles with the “black” phenotype characteristic of East Pacific populations are sometimes encountered in Hawaii and even the west Pacific, but these appear to be rare cases of feeding dispersal of juveniles, and no movement between Hawaiian and E. Pacific breeding sites has been documented.
Significance

Although Hawaii is better known for its high degree of endemism in birds, plants, and Drosophila (fruit flies), substantial fractions (20–45 percent) of its species in many marine taxa are also only found only in Hawaii (Briggs, 1974). The distinctive marine biota, together with the unusual and diverse habitats along the island chain, create a unique ecological setting for green turtles that is not duplicated anywhere else in the world. This DPS has no continental-shelf habitats, a characteristic shared only with the Central South Pacific DPS. If all turtles were lost from this vast geographic area, it would create a major gap in the global range of the species.

4.2.11. East Pacific DPS

Discreteness

The North and South American continents bound this DPS to the east, while the East Pacific Barrier, an uninterrupted 4,000 mile stretch of water with depths up to 7 miles, largely restricts movements of turtles between this DPS and others in the Pacific Ocean. Turtles from the East Pacific carry mtDNA haplotypes from a clade that differs from those in the Central South Pacific DPS, which indicates essentially complete isolation over evolutionary time scales. Turtles from Hawaii have mtDNA from the same clade, but the array of haplotypes is almost completely non-overlapping between Hawaii and East Pacific, indicating a substantial degree of ongoing isolation between the two areas. Tagged juvenile turtles from the East Pacific have been recovered in the western Pacific, but these are believed to represent feeding migrations rather than reproduction. No satellite-tagged adults have dispersed to areas outside the DPS, nor have satellite-tracked turtles from elsewhere migrated into the East Pacific.

Significance

The two cold-water currents on the east side of the Pacific Ocean (the Humboldt Current in the south and the California Current in the north) leave a distinctive region of tropical ocean along the west coasts of Mexico, Central America, and northern South America that is known as the Eastern Pacific Zoogeographic Region (Briggs, 1974). East Pacific turtles exhibit marked genetic separation from all other DPSs, with the possible exception of the Central North Pacific DPS. Mean size of nesting turtles in the East Pacific is smaller than in any other DPS, which could reflect an adaptation to local ecological conditions, as could the distinctive “black” phenotype. Loss of all turtles from this DPS would leave a gap in the range along the entire eastern boundary of the world’s largest ocean.
5. NORTH ATLANTIC DPS (DPS #1)

5.1. DPS Range and Nesting Distribution

The North Atlantic DPS extends from the boundary of South and Central America north to the northern extent of the green turtle’s range to include Panama, Costa Rica, Nicaragua, Honduras, Belize, Mexico, and the United States; due east across the Atlantic Ocean, and south to the Islamic Republic of Mauritania (Mauritania) on the African continent; and west to the Caribbean basin, including Puerto Rico, the Bahamas, Cuba, Turks and Caicos Islands, Republic of Haiti (Haiti), Dominican Republic, Cayman Islands, and Jamaica (Figure 5.1).

Figure 5.1. Geographic range of the North Atlantic DPS. Size of circles indicates estimated nester abundance (see Section 5.2.1) nesting abundance category. Locations marked with ‘×’ indicate nesting sites lacking abundance information.

Four regions support nesting concentrations of particular interest in the North Atlantic DPS: Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and Quintana Roo); U.S. (Florida), and Cuba (Figure 5.2). By far the most important nesting concentration for green turtles in this DPS is Tortuguero, Costa Rica. In Mexico, nesting occurs primarily along the Yucatan Peninsula, with lower nesting densities in Tamaulipas and Veracruz. In Florida, nesting occurs in coastal areas of all regions except the Big Bend area of west central Florida; however, the bulk of nesting occurs along the Atlantic coast of eastern central Florida. In Cuba, nesting primarily occurs on the extreme western tip of the country (Guahanacabibes Peninsula) and islands to the south (San Felipe Keys, Canareeos Archipelago, and Jardines de la Reina Archipelago). Nesting also occurs in the Bahamas, Belize, Cayman Islands, Dominican Republic, Haiti, Honduras,
Jamaica, Nicaragua, Panama, Puerto Rico, Turks and Caicos Islands, and North Carolina, South Carolina, Georgia, and Texas, U.S.A. In the eastern North Atlantic, nesting has been reported in Mauritania (Fretey, 2001).

Figure 5.2. Close up of nesting distribution of green turtles in the western North Atlantic DPS. Size of circles indicates estimated nester abundance (see Section 5.2.1) nesting abundance category. Locations marked with ‘×’ indicate nesting sites lacking abundance information.

Green turtle neritic foraging grounds in the North Atlantic range from coral or nearshore reefs and seagrass beds, to inshore bays and estuaries (Ehrhart, 1983; Guseman and Ehrhart, 1990; Wershoven and Wershoven, 1992; Bresette et al., 1998; Schroeder et al., 1998; Bresette et al., 2002; Ehrhart et al., 2007), to man-made embayments (Redfoot and Ehrhart, 2000; Kubis et al., 2009). Turtles feed primarily on sea grass or benthic macroalgae depending on the habitat in which they reside (Bjorndal, 1980; Mortimer, 1981; Coyne, 1994; Shaver, 1994; Redfoot, 1997; Vander Zanden et al., 2013a), and they change habitats during successive stages of life (Reich et al., 2008; Bagley et al., 2008, Vander Zanden et al., 2013b). In the western North Atlantic, juvenile green turtles forage as far north as Cape Cod Bay, Massachusetts, as far east as Bermuda, and throughout the Caribbean; however, foraging adults are only found from the southernmost reach of the Florida peninsula south (Witherington et al., 2006). In the eastern North Atlantic, juvenile green turtles are present year round in Mauritania (Fretey, 2001), and occur occasionally in the waters of the Azores, Madeira (Groombridge and Luxmoore, 1989 as cited in Fretey, 2001), the Kingdom of Morocco (Morocco; de los Rios y Loshuertos et al., 2008), and the Canary Islands (Machado, 1989 as cited in Fretey, 2001). An important foraging ground in the North Atlantic DPS for nesters from the South Atlantic DPS (Poilão, Guinea Bissau) is the Parc National du Banc d’Arguin in Mauritania (Godley et al., 2003).
5.2. **Critical Assessment Elements**

In the evaluation of extinction risk for green turtles in the North Atlantic DPS, the SRT considered six critical assessment elements, each of which are discussed below: (1) Nesting Abundance, (2) Population Trends (3) Spatial Structure, (4) Diversity / Resilience, (5) Five-Factor Threat Analyses, and (6) Existing Conservation Efforts. See Section 3.3 for additional information on the selection of these six critical assessment elements.

5.2.1. **Nesting Abundance**

The SRT identified 76 nesting sites within the North Atlantic DPS, although some represent numerous individual beaches. There are four regions that support high density nesting concentrations for which data were available: Costa Rica (Tortuguero), Mexico (Campeche, Yucatan, and Quintana Roo), U.S. (Florida), and Cuba (Table 5.1). Nester abundance was assessed by the SRT for 48 nesting sites within the North Atlantic DPS. Abundance was estimated using the best scientific information available. Remigration intervals and clutch frequencies were used to estimate total nester abundance when counts of nesters were not available using the following equation: 

\[
\text{Adult Female Abundance} = \frac{\text{nests}}{\text{clutch frequency}} \times \text{remigration interval}
\]

In terms of nester distribution, the largest nesting site (Tortuguero, Costa Rica) hosts 79 percent of total nester abundance (167,528 nesters; Table 5.2). There were also 26 nesting sites for which we have qualitative reports of nesting activity but no nesting data: 3 in the Bahamas, 3 in Belize, 1 in Costa Rica, 4 in Cuba, 1 in the Dominican Republic, 1 in Haiti, 6 in Honduras, 2 in Jamaica, 1 in Mauritania, 1 in Panama, and 3 in the Turks and Caicos Islands.
Table 5.1. Summary of green turtle nesting sites in the North Atlantic DPS. Data are organized by country, nesting site, monitoring period, and estimated total nester abundance \([\text{total counted females / years of monitoring}] \times \text{remigration interval}\), and represent only those sites with sufficient data to estimate number of females. For a list of references for these data, see Appendix 2.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cayman Islands</td>
<td>Grand Cayman</td>
<td>2005-2009</td>
<td>72</td>
</tr>
<tr>
<td>Cayman Islands</td>
<td>Little Cayman</td>
<td>2007</td>
<td>5</td>
</tr>
<tr>
<td>Cuba</td>
<td>Cayo Largo (Eastern Keys of Isla de la Juventud)</td>
<td>2001-2010/2008-2010</td>
<td>1,284</td>
</tr>
<tr>
<td>Cuba</td>
<td>Beaches of the Guanacabibes Peninsula</td>
<td>1998-2010/2010-2012</td>
<td>201</td>
</tr>
<tr>
<td>Cuba</td>
<td>South Isla de la Juventud</td>
<td>2010-2011</td>
<td>170</td>
</tr>
<tr>
<td>Cuba</td>
<td>San Felipe</td>
<td>2004-2011</td>
<td>162</td>
</tr>
<tr>
<td>Cuba</td>
<td>Cayo Siju, Cayo Real, Juan Garcia (Cayos de San Felipe)</td>
<td>2007-2009</td>
<td>123</td>
</tr>
<tr>
<td>Cuba</td>
<td>Playas Archipielago Jardines de la Reina</td>
<td>2011</td>
<td>88</td>
</tr>
<tr>
<td>Cuba</td>
<td>Eastern Keys of Isla de la Juventud</td>
<td>2010</td>
<td>64</td>
</tr>
<tr>
<td>Cuba</td>
<td>Cayo Rosario</td>
<td>2008</td>
<td>10</td>
</tr>
<tr>
<td>Mexico</td>
<td>Quintana Roo</td>
<td>2010-2012</td>
<td>18,257</td>
</tr>
<tr>
<td>Mexico</td>
<td>Campeche</td>
<td>1992-2010</td>
<td>2,207</td>
</tr>
<tr>
<td>Mexico</td>
<td>Yucatan</td>
<td>2006-2011</td>
<td>2,111</td>
</tr>
<tr>
<td>Mexico</td>
<td>Veracruz</td>
<td>2004-2006/1998-2000</td>
<td>1,040</td>
</tr>
<tr>
<td>Mexico</td>
<td>Tamaulipas</td>
<td>2009-2010</td>
<td>715</td>
</tr>
<tr>
<td>Nicaragua</td>
<td>El Cocal</td>
<td>2000</td>
<td>6</td>
</tr>
<tr>
<td>Puerto Rico*</td>
<td>Vieques</td>
<td>2010-2012</td>
<td>626</td>
</tr>
<tr>
<td>Puerto Rico*</td>
<td>Mona Island</td>
<td>2012</td>
<td>12</td>
</tr>
<tr>
<td>Puerto Rico*</td>
<td>Humacao</td>
<td>2012</td>
<td>6</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Brevard County</td>
<td>2008-2012/2011-2012</td>
<td>3,979</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Palm Beach County</td>
<td>2009-2010/2011-2012</td>
<td>2,006</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Martin County</td>
<td>2008-2012/2011-2012</td>
<td>998</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Indian River County</td>
<td>2008-2012/2011-2012</td>
<td>504</td>
</tr>
<tr>
<td>USA, FL</td>
<td>St. Lucie County</td>
<td>2008-2012/2011-2012</td>
<td>233</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Volusia County</td>
<td>2008-2012/2011-2012</td>
<td>215</td>
</tr>
</tbody>
</table>

Comment [A8]: So, this could potentially include counts of the same individual that may have nested multiple times over many years? If so, then these data are potentially over-estimates...perhaps average annual nester estimates would be a better proxy??
Response: Because remigration interval is part of the computation, we believe this accounts for seeing the same individual. It is not perfect, but our best estimate.

Comment [A9]: Error/range?
Response: No statistics were performed; see section 5.2.1 for explanation of abundance computation.
<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>USA, FL</td>
<td>Broward County</td>
<td>2008-2012/2011-2012</td>
<td>157</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Monroe County</td>
<td>2008-2012/2011-2012</td>
<td>120</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Dry Tortugas National Park</td>
<td>2009-2010</td>
<td>104</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Flagler County</td>
<td>2008-2012/2011-2012</td>
<td>39</td>
</tr>
<tr>
<td>USA, NC</td>
<td>North Carolina</td>
<td>2010-2012/2011-2012</td>
<td>39</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Sarasota County</td>
<td>2008-2012/2011-2012</td>
<td>21</td>
</tr>
<tr>
<td>USA, FL</td>
<td>St. Johns County</td>
<td>2008-2012/2011-2012</td>
<td>20</td>
</tr>
<tr>
<td>USA, TX</td>
<td>Texas</td>
<td>2010-2012/2011-2012</td>
<td>16</td>
</tr>
<tr>
<td>USA, SC</td>
<td>South Carolina</td>
<td>2010-2012/2011-2012</td>
<td>11</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Lee County</td>
<td>2008-2012/2011-2012</td>
<td>9</td>
</tr>
<tr>
<td>USA, GA</td>
<td>Georgia</td>
<td>2011/2012</td>
<td>5</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Miami-Dade County</td>
<td>2008-2012/2011-2012</td>
<td>5</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Charlotte County</td>
<td>2008/2011-2012</td>
<td>3</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Escambia County</td>
<td>2008/2011-2012</td>
<td>3</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Collier County</td>
<td>2008/2011-2012</td>
<td>2</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Nassau County</td>
<td>2008/2011-2012</td>
<td>2</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Okaloosa County</td>
<td>2008/2011-2012</td>
<td>2</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Duval County</td>
<td>2008/2011-2012</td>
<td>1</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Franklin County</td>
<td>2008/2011-2012</td>
<td>1</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Manatee County</td>
<td>2008/2011-2012</td>
<td>1</td>
</tr>
<tr>
<td>USA, FL</td>
<td>Walton County</td>
<td>2008/2011-2012</td>
<td>1</td>
</tr>
</tbody>
</table>

* These sites were added to the table following the votes on the critical assessment elements and the probability of reaching a critical risk threshold, and thus were not considered in these votes (see section 5.3). However, because they represent only 0.4% of the estimated nester abundance and were within a DPS portion already containing major nesting sites, we don’t consider them to be of sufficient significance to trigger a new round of extinction risk voting.
Table 5.2. Green turtle nester abundance distribution among nesting sites in the North Atlantic DPS.

<table>
<thead>
<tr>
<th>NESTER ABUNDANCE</th>
<th># NESTING SITES DPS 1</th>
</tr>
</thead>
<tbody>
<tr>
<td>n/a</td>
<td>26</td>
</tr>
<tr>
<td>1–10</td>
<td>17</td>
</tr>
<tr>
<td>11–50</td>
<td>6</td>
</tr>
<tr>
<td>51–100</td>
<td>3</td>
</tr>
<tr>
<td>101–500</td>
<td>10</td>
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<tr>
<td>501–1000</td>
<td>4</td>
</tr>
<tr>
<td>1001–5000</td>
<td>6</td>
</tr>
<tr>
<td>5001–10000</td>
<td>0</td>
</tr>
<tr>
<td>10001–100000</td>
<td>1</td>
</tr>
<tr>
<td>&gt;100,000</td>
<td>1</td>
</tr>
<tr>
<td>Total Sites</td>
<td>74</td>
</tr>
<tr>
<td>Total Abundance</td>
<td>167,528</td>
</tr>
</tbody>
</table>
| PERCENTAGE at Largest NESTING SITE | 79%  
(Tortuguero, Costa Rica)

5.2.2. Population Trends

Green turtle nesting populations in the North Atlantic are some of the most studied in the world, with time series exceeding 40 years in Costa Rica and 35 years in Florida. For a list of references on trend data, see Appendix 3.

There are seven sites for which 10 years or more of recent data are available for annual nester abundance (the standards for representing trends in bar plot in this report; Figure 5.3). Of these, four sites met our standards for conducting a PVA, and thus are not represented in the bar plots below. See Section 3.2 for more on data quantity and quality standards used for bar plots and PVAs.

Comment [A10]: 35 years
Response – we used 20 years to reflect the INBS time series, but agree that 35 is more appropriate.

Comment [A11]: Then why do we only show the last 3-5 years in table 5.1?
Response: To calculate nester abundance, we used the latest (most recent) information available. For Florida, since the remigration interval is 2 years, we only used the most recent 2 years of data. For other sites with longer remigration intervals, 3 years of data were used.

Comment [A12]: Comment: GEORGIA, SOUTH CAROLINA AND NORTH CAROLINA HAVE 10+ YEARS OF MONITORING FOR GREEN TURTLES, GOING BACK FROM 2013... PERHAPS OTHERS.
Response: Due to the low number of green turtle nesting in these States, these sites were not included to analyze trends.
Figure 5.3. Trend data for green turtle nesting in the North Atlantic DPS with greater than 10 yrs of recent monitoring data, with a missing year. These include El Cuyo, Mexico (14 yrs), San Felipe, Cuba (11 yrs), and Guanal, Cuba (14 yrs).

Of the three sites with bar plots, there were apparent patterns of hi-low nesting in El Cuyo, Mexico and Guanal, Cuba with the exception of 2003 and 2004 nesting season in El Cuyo, Cuba. No trend was detected for these sites. In San Felipe, Cuba, the last two years of nesting were higher than previous years.

Elsewhere in Mexico, especially along the Yucatan Peninsula, nesting has also increased. In the early 1980s, approximately 875 nests/yr were deposited, but by 2000 this increased to over 1,500 nests/year (NMFS and USFWS, 2007). In 2012, more than 26,000 nests were deposited in Quintana Roo (J. Zurita, CIQROO, unpubl. data, 2013). The estimated total nester abundance for Mexico (in 5 states: Campeche, Quintana Roo, Tamaulipas, Veracruz, Yucatan) is 24,330 turtles.

PVA was one aspect of the Population Trend element and was conducted for nesting sites that had a minimum of 15 years of recent nesting data with an annual nesting level of more than 10 females (for more on data quantity and quality standards used, see Section 3.2). There were four nesting sites that met these criteria: Tortuguero, Costa Rica; Isla Aguada, Mexico; Guanahacabibes, Cuba; and Florida, U.S.A. To assist in interpreting these PVAs, we indicate the probability of green turtle nesting populations declining to two different biological reference points, one using a trend-based, and the other an abundance-based threshold. The trend-based reference point for evaluating population forecasts is half of the last observed abundance value,
i.e., a 50 percent decline. The abundance-based reference point was a total adult female abundance of 300 females. Risk is calculated as the percentage of model runs that fall below these reference points after 100 years. It should be noted that this PVA modeling has important limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. For a full discussion of these PVAs and these reference points, see Section 3.2.

Tortuguero, Costa Rica is the most important nesting concentration for green turtles in this DPS (Figure 5.4). This population has been studied since the 1950s and nesting has increased markedly since the early 1970s. From 1971 to 1975, there were approximately 41,250 nesting emergences per year and from 1992 to 1996 there were approximately 72,200 nesting emergences per year (Bjorndal et al., 1999). From 1999 to 2003, about 104,411 nests/year were deposited, which corresponds to approximately 17,402-37,290 nesting females each year (Troëng and Rankin, 2005). An estimated 180,310 nests were laid during 2010, the highest level of green turtle nesting estimated since the start of nesting track surveys in 1971. This equates to 30,052-64,396 nesters in 2010. This increase has occurred despite substantial human impacts to the population at the nesting beach and at foraging areas (Troëng, 1998; Campbell and Laguseux, 2005; Troëng and Rankin, 2005).

Figure 5.4. Stochastic Exponential Growth (SEG) Model Output at Tortuguero, Costa Rica. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference. Number of nesters was computed from nests using 2.8 nests per female (Tortuguero, Costa Rica; Carr et al., 1978).

The Costa Rica analysis was completed using an index of adult female nesters across 41 seasons from 1971 to 2011 (Figure 5.4). Nesting beach monitoring data and the PVA indicate that there is a 0.7 percent probability that this population will fall below the trend reference point (50 percent decline) at the end of 100 years, and a 0 percent probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years.

Comment [A15]: This is the value in the literature, but it is low in comparison to other populations. Because these rates tend to be underestimates, all the clutch frequencies may be low. Variation between populations is likely an illusion, and is based on differing techniques and effort, not on biology. The SRT might consider applying the most trustworthy estimate within a DPS to all populations within the DPS.

Response: We agree with your statement; however, the SRT decided to use the available published literature for each site. If no data were available, then data from the closest neighboring population was used.
The Isla Aguada, Mexico analysis (Figure 5.5) was completed using an index of adult female nesters across 21 seasons from 1992 to 2011 based on data from Guzmán-Hernández and García Alvarado (2013; 2012; 2011; 2010; 2009), Guzmán-Hernández et al. (2008), and Guzmán- Hernández (2006a; 2006b; 2005; 2003; 2002; 2001; 2000).

Figure 5.5. Stochastic Exponential Growth (SEG) Model Output for Isla Aguada, Mexico. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference. Number of nesters was computed from nests using 4.01 nests per female.

Nesting beach monitoring data and the PVA indicate that there is a 3.7 percent probability that this population will fall below the trend reference point (50 percent decline) at the end of 100 years, and a 2.2 percent probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years.

The Guanahacabibes, Cuba analysis was completed using an index of adult female nesters across 15 seasons from 1989 to 2012 based on data from Azanza-Ricardo, 2009 and Azanza-Ricardo et al., 2013 (Figure 5.6). The units for the Guanahacabibes indices are expressed as adult females, so no transformation from nests to nesters was needed.
Nesting beach monitoring data indicate that there is a 27.8 percent probability that this population will fall below the trend reference point (50 percent decline) at the end of 100 years, and the model estimates a 37.3 percent probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years.

Beaches of the Guahanacabibes Peninsula, Cuba have been monitored for nesting activity since 1999. Cayo Largo in the Canarreos Archipelago hosts the largest nesting population in Cuba with an estimated 1,284 females (SWOT, 2013). Other nest monitoring programs occur on the San Felipe Keys, Canarreos Archipelago, Jardines de la Reina Archipelago, and Guanal. The estimated total nester abundance for nesting sites for which we had data is 2,226 turtles.

The analysis for Florida, U.S.A. was conducted using an index of adult female nesters across 24 seasons from 1989 to 2012 based on data from index nesting beaches (FWC, 2013; Figure 5.7).
Figure 5.7. Stochastic Exponential Growth (SEG) Model Output for Florida Index Beaches. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference. Number of nesters was computed from nests using 3 nests per female Index beaches, FL; Johnson and Ehrhart, 1996).

Nesting beach monitoring data and the PVA indicate that there is a 0.3 percent probability that this population will fall below the trend reference point (50 percent decline) at the end of 100 years, and a 0 percent probability that this population will fall below the absolute abundance reference (100 females per year) within 100 years.

In Florida, nesting occurs in coastal areas of all regions except the Big Bend area of west central Florida. The bulk of nesting occurs along the Atlantic coast of eastern central Florida, where a mean of 5,055 nests were deposited each year from 2001 to 2005 (Meylan et al., 2006) and 10,377 each year from 2008 to 2012 (B. Witherington, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013). Nesting has increased substantially over the last 20 years and peaked in 2011 with 15,352 nests statewide (Chaloupka et al., 2008; B. Witherington, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013). The estimated total nester abundance for Florida is 8,426 turtles.

Similar to the nesting trend found in Florida, in-water studies in Florida have also recorded an increasing trend in green turtle captures at the Indian River Lagoon site, with a 661 percent increase over 24 years (Ehrhart et al., 2007), and the St Lucie Power Plant site, with a significant increase in the annual rate of capture of immature green turtles (SCL<90 cm) from 1977 to 2002 or 26 years (3,557 green turtles total; M. Bressette, Inwater Research Group, unpubl. data; Witherington et al., 2006).
5.2.3. Spatial Structure

When examining spatial structure for the North Atlantic DPS, the SRT examined three lines of evidence including genetic data, flipper and satellite tagging data, and demographic data.

Genetic sampling in the North Atlantic DPS has been generally extensive with good coverage of large populations in this region; however, some smaller Caribbean rookeries are absent and coastal rookeries in the Gulf of Mexico are under-represented. Genetic differentiation based on mtDNA indicated that there are at least 4 independent nesting subpopulations in the North Atlantic DPS characterized by shallow regional substructuring: (1) Florida (Hutchinson Island), (2) Cuba (Guanahacabibes Peninsula and Cayeria San Felipe), (3) Mexico (Quintana Roo), and (4) Costa Rica (Tortuguero; Lahanas et al., 1994, Ruiz-Urquiola et al., 2010). These rookeries are characterized by common and widespread haplotypes dominated by CM-A1 and/or CM-A3. A relatively low level of spatial structure is detected due to shared common haplotypes, although there are some rare/unique haplotypes at some rookeries. Connectivity may indicate recent shared common ancestry. Studies using nuclear DNA markers raised the possibility of connectivity via male-mediated gene flow among Atlantic nesting populations. In particular, they highlighted the degree of mixing between nesting populations on Suriname and Ascension Island, which are distinct maternally (mtDNA) but share a feeding ground along the coast of Brazil (Karl et al., 1992; Roberts et al., 2004).

Mixed-stock analysis of foraging grounds show that green turtles from multiple nesting beaches commonly mix at feeding areas across the Caribbean and Gulf of Mexico, with higher contributions from nearby large nesting sites and some contribution estimated from nesting populations outside the DPS (Bass et al., 1998; Bass and Witzell 2000; Bolker et al., 2007; Bjorndal and Bolten 2008). There is, however, an overlap in foraging areas between the eastern and western Caribbean rookeries (DPS 1 and 3). Lahanas et al. (1998) showed that juvenile green turtles in the Bahamas originate mainly from western Caribbean (Tortuguero, Costa Rica) (79.5 percent) but that a significant proportion may be coming from the eastern Caribbean (Aves Island/Suriname; 12.9 percent). There is evidence that dispersal of juveniles from nesting populations in the South Atlantic (DPS 3) to the North Atlantic (DPS 1) is limited (Bass et al., 2004; Bolker et al., 2007).

There are several sites in the North Atlantic DPS with long-term flipper and/or satellite tagging projects. Flipper tagging studies on foraging grounds and/or nesting beaches have been conducted in Bermuda (Meylan and Meylan, 2011), Costa Rica (Troêng et al., 2005), Cuba (Moncada et al., 2006), Florida (Johnson and Ehrhart, 1996; Kubis et al., 2009), Mexico (Zurita et al., 2003; 1994), Panama (Meylan and Meylan, 2011), Puerto Rico (Collazo et al., 1992; Patricio et al., 2011)), and Texas (Shaver, 2002; 1994). Nesters have been satellite tracked from Florida, Cuba, Cayman Islands, Mexico, and Costa San. While there is some crossover into the South Atlantic DPS from nesters in the equatorial region, North Atlantic DPS nesters primarily reside in foraging beds pastures within the DPS (Troêng et al., 2005).

Green turtles in the neritic and oceanic zones in Florida waters have been studied to various extents in areas where inwater research projects have been conducted (Eaton et al., 2008). These research projects include tracking post-hatchling to adult green turtles. Post-hatchlings and juvenile turtles monitoring and research in Florida: review and recommendations. U.S. Dep. Commerce, NOAA Tech. Memo. NMFS-OPR-38, 233 p.
pelagic green turtles are found mostly along the central and northern east coast associated with Sargassum (Witherington et al., 2006). Juvenile green turtles then return to neritic waters such as nearshore reefs and coastal lagoons, such as the Indian River Lagoon (Mendonça and Ehrhart, 1982; Ehrhart et al., 2007; Makowski et al., 2006) and St. Joseph Bay in the Florida. Other developmental habitats where juvenile green turtles have been recorded include the Trident Submarine Basin at Port Canaveral [Redfoot and Ehrhart, 2013] and the Cape Canaveral Submarine Basin at Port Canaveral [Redfoot and Ehrhart, 2013] and the Cape Canaveral Shipping Channel [Henwood and Ogren, 1987]. Larger juveniles and adult green turtles have been found foraging in the Florida Keys [Bresette et al., 2010]. Foraging areas outside Florida for these size classes include the Caribbean and Bahamas [Ehrhart et al., 2007] Bagley et al., 2008]. Post-nesting green turtles, that have been satellite tracked, forage in areas near to the Dry Tortugas and the Bahamas (Witherington et al., 2006). Green turtles from Cuba, Costa Rica, and the Cayman Islands have been found in Florida waters [Troëng et al., 2005] Blumenthal et al., 2006], and Cuba [Moncada et al., 2006].

The demography of green turtles in the North Atlantic DPS appears to be consistent among the various nesting assemblages. This consistency in parameters such as mean nesting size, internesting interval, clutch size, hatching success, nesting season, and clutch frequency suggests a low level of population structuring in the North Atlantic DPS.

Size of nesters ranges from 101.7 cm CCL (Campeche, Mexico) to 109.3 cm CCL (Isla Aguada, Mexico (Guzmán-Hernández, 2001, 2006a). The internesting interval ranges from 9 to 18 days (Witherington and Ehrhart, 1989; Johnson and Ehrhart, 1996; Troëng et al., 2005; Hart et al., 2013) and on average, females lay 3 clutches per season (range estimated from 2.8 to 4.6 nests per season; Guzmán-Hernández and Garcia, 2013; 2012; 2011; 2010; 2009; Johnson and Ehrhart, 1996; Carr et al., 1978). Remigration intervals have been reported between 2 and 3 years (Troëng and Chaloupka, 2007; Zurita et al., 1994; Witherington and Ehrhart, 1989). Furthermore, green turtle clutches range from 108 eggs in Costa Rica (Tiwari et al., 2006) to 136 eggs in Florida (Witherington and Ehrhart, 1989), and have a hatching success ranging from 61.6 percent in Florida (Witherington and Ehrhart, 1989) to 92 percent in Mexico (Xavier et al., 2006) to 97 percent in Mexico (Xavier et al., 2006), although the high of 92 percent is an overestimate should be much lower since nests that did not show signs of hatching were excluded from the analysis.

Age at first reproduction is known for 2 sites: 12-26 at Tortuguero, Costal Rica (Frazer and Ladner, 1986) and 12-20 with an average of 16 at Quintana Roo, Mexico (Richards et al., 2011).

5.2.4 Diversity / Resilience

The components considered under this critical element include the spatial range of nesting sites, diversity in nesting season, site structure, orientation (e.g., high vs. low beach face, insular vs. continental nesting sites), and the genetic diversity within the DPS. These are important considerations for assessing the potential impact of catastrophic events such as storms, sea level rise, and disease.

The overall nesting range for the North Atlantic DPS is vast. Green turtles nest on both continental and island beaches throughout the DPS (Witherington et al., 2006). Major nesting
sites are primarily continental with hundreds of lower density sites scattered throughout the Caribbean. Green turtles nesting in Florida seem to prefer barrier island beaches that receive high wave energy and that have coarse sands, steep slopes, and prominent foredunes. The greatest nesting is on sparsely developed beaches that have minimal levels of artificial lighting. Green turtles typically deposit their eggs near the base of the primary dune (Witherington et al., 2006). Green turtles select beaches with slightly steeper slopes than hawksbills at El Cuyo, Mexico (Cuevas et al., 2010). The high-low nesting pattern for Florida and Mexico occurs during the same years; however, nesting in Tortuguero, Costa Rica is not always in sync with Florida and Mexico (e.g., 2011 was a high nesting year in Florida, but for Tortuguero the high nesting year was 2010). The nesting season is similar throughout the DPS, with green turtles nesting from June to November in Costa Rica (Bjorndal et al., 1999), and May through September in the U.S., Mexico, and Cuba (Witherington et al., 2006). The fact that turtles nest on both insular and continental sites suggests a high degree of nesting diversity.

Mitochondrial DNA studies have identified at least 4 independent nesting subpopulations characterized by shallow regional sub structuring (Encalada et al., 1996; Ruiz-Urquiola et al., 2010). Identified genetic stocks are (1) U.S. (Hutchinson Island, Florida), (2) Cuba (Guanahacabibes peninsula and Cayería San Felipe), (3) Mexico (Quintana Roo), and (4) Costa Rica (Tortuguero).

5.2.5. Analysis of Factors Listed Under ESA Section 4(a)(1)

Section 4(a)(1) of the ESA lists five factors (threats) that must be considered when determining the status of a species. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

The following information on these factors /threats pertains to green turtles found in the North Atlantic DPS.

5.2.5.1. Factor A: Destruction or Modification of Habitat or Range

Coastal development, beachfront lighting, erosion resulting from sand mining, non-native vegetation, and sea level rise resulting from climate change all negatively affect hatchlings and nesting turtles throughout this DPS. Fishing practices and marine pollution also affect the turtles throughout the DPS, with higher numbers of interactions occurring in waters where green turtles are known to forage and migrate. All life stages of green turtles are affected by habitat destruction in the neritic/oceanic zone.
**Terrestrial Zone**

In the North Atlantic DPS, some nesting beaches continue to be severely degraded from a variety of activities. Destruction and modification of green turtle nesting habitat results from coastal development, construction, beachfront lighting, placement of erosion control structures and other barriers to nesting, placement of nearshore shoreline stabilization structures, vehicular and pedestrian traffic, beach erosion, beach sand placement, removal of native vegetation, and planting of non-native vegetation.

Numerous beaches in the North Atlantic DPS are eroding due to both natural (e.g., storms, sea level changes, waves, shoreline geology) and anthropogenic (e.g., construction of armoring structures, groins, and jetties; marinas; coastal development; inlet dredging) factors. Such shoreline erosion leads to a loss of nesting habitat for green turtles.

The beaches of the Atlantic coast of Mexico are threatened by habitat loss and degradation due to coastal development, sand mining, structures on the beach such as geotubes, and trash on the beach. Nests on Isla Contoy in the Yucatan Peninsula, Mexico, and Isla Aguada in Campeche, Mexico, are susceptible to increased erosion by regular high tides, storm events (Duran Najera, 1990; Guzmán-Hernández and García Alvarado, 2011). Isla Contoy and Isla Holbox have been declared Ecological Reserves. Fortunately, Isla Contoy is protected from coastal development (Hildebrand, 1987); however, on the Mexican mainland at Rio Lagartos, a Biosphere Reserve, the salt extraction industry affects the adjacent coastal dunes and causes further loss of habitat (Duran Najera, 1990). On the beaches of Alto Lucero in Veracruz, Mexico, loss of nesting habitat is due to erosion caused by hurricanes, strong rains, and big tides (Dirado et al., 2002). There is an increase in development on the coastline of Alto Lucero which has increased the level of artificial lighting in the area and thus decreased the quality of nesting habitat. Three kilometers of the coastline of beaches are considered unusable to nesting green turtles due to the brightly illuminated Laguna Verde nuclear power plant (Dirado et al., 2000). In Quintana Roo, Mexico, the main threats to the nesting habitat are coastal development and tourism (Zurita et al., 1993).

In the southeastern United States, numerous erosion control structures that create barriers to nesting have been constructed. The proportion of coastline that is armored is approximately 18 percent (239 km) in Florida (Clark, 1992; Schroeder and Mosier, 2000; Witherington et al., 2006; Witherington et al., 2011). These assessments of armoring extent do not include structures that are a barrier to sea turtle nesting but that do not fit the definition of armoring, such as dune crossovers, cabanas, sand fences, and recreational equipment. Jetties have been placed at many ocean inlets in the southeastern United States, to keep transported sand from closing the inlet channel. The effect of inlets in lowering sea turtle nesting density was observed both 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). There are some efforts, such as the Coastal Construction Control Line Program (CCCL), that provide protection for Florida's beaches and dunes while allowing for continued use of private property. The CCCL program establishes a coastal construction control line in which special siting and design criteria are applied for construction and related activities.
Armoring structures on and adjacent to the nesting beach continue to be permitted on the nesting beaches of Florida.

Also in the southeastern United States, beach nourishment is a frequent activity, and many beaches are on a periodic nourishment schedule. On severely eroded sections of beach, where little or no suitable nesting habitat previously existed, beach nourishment has been found to result in increased nesting (Ernest and Martin, 1999). However, on most beaches in the southeastern United States, nesting success typically declines for the first year or two following construction, even though more nesting habitat is available for turtles (Trindell et al., 1998; Ernest and Martin, 1999; Herren, 1999; Brock et al., 2009). Reduced nesting success on constructed beaches has been attributed to increased sand compaction, escarpment formation, and changes in beach profile (Nelson et al., 1987; Crain et al., 1995; Lutcavage et al., 1997; Steinitz et al., 1998; Ernest and Martin, 1999; Rumbold et al., 2001; Brock et al., 2009). Stormwater and other water source runoff from beachfront parking lots, building rooftops, roads, decks, and draining swimming pools adjacent to the beach is frequently discharged directly onto southeastern U.S beaches and dunes either by sheet flow, through stormwater collection system outfalls, or through small diameter pipes. These outfalls create localized erosion channels, prevent natural dune establishment, and wash out sea turtle nests (NMFS and FWS, 2008). Contaminants contained in stormwater, such as oils, grease, antifreeze, gasoline, metals, pesticides, chlorine, and nutrients, are also discharged onto the beach.

In Florida, vehicular driving is allowed on certain beaches along the northeast coast (Nassau, Duval, St. Johns, and Volusia Counties) and the northwest coast (Walton and Gulf Counties). Driving also occurs in Georgia (Cumberland, Little Cumberland, and Sapelo Islands), North Carolina (Fort Fisher State Recreation Area, Carolina Beach, Freeman Park, Onslow Beach, Emerald Isle, Indian Beach/Salter Path, Pine Knoll Shores, Atlantic Beach, Cape Lookout National Seashore, Cape Hatteras National Seashore, Nag’s Head, Kill Devil Hills, Town of Duck, and Currituck Banks), and Texas (the majority of beaches except for a highly developed section of South Padre Island and Padre Island National Seashore, San Jose Island, Matagorda Island, and Matagorda Peninsula where driving is not allowed or is limited to agency personnel, land owners, and/or researchers). However, green turtles nest in much smaller numbers in Georgia, North Carolina, and Texas than they do in Florida; thus, impacts to green turtle nesting habitat is not as significant in these States. Beach driving has been found to reduce the quality of green turtle nesting habitat in several ways. In the southeastern U.S., vehicle ruts on the beach have been found to 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). Vehicle lights and vehicle movement on the beach after dark results in reduced habitat suitability, which 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.

In Florida, green turtle nesting habitat is under constant threat from coastal development and other forms of disruptive human activity (Witherington and Koeppel, 2000). Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and
 Loss of nesting habitat related to coastal development has had the greatest impact on nesting sea turtles in Florida. Beachfront development not only causes the loss of suitable nesting habitat, but can result in the disruption of powerful coastal processes accelerating erosion and interrupting the natural shoreline migration (National Research Council, 1990). This may in turn cause the need to protect upland structures and infrastructure by armoring, groin placement, beach emergency berm construction and repair, and beach nourishment. All of these beach stabilization measures cause changes in, additional loss of, or other impacts to the remaining sea turtle habitat. These factors may directly, through loss of beach habitat, or indirectly, through changing thermal profiles and increasing erosion, serve to decrease the amount of nesting area available to nesting females, and may evoke a change in the natural behaviors of adults and hatchlings (Ackerman, 1997; Schroeder and Mosier, 2000).

In addition, coastal development is usually accompanied by artificial lighting. The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington, 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Witherington and Bjorndal, 1991; Nelson Sella et al., 2006). These threats have been well documented along the coastal stretches of Florida. Based on hatching orientation index surveys at nests located at 23 representative beaches in six counties around Florida in 1993 and 1994, Witherington et al. (1996) found that, by county, approximately 10 to 30 percent of all sea turtle nests showed evidence of hatchlings disoriented by lighting. Changing to sea turtle compatible lighting has been accomplished at the local level through voluntary compliance or by adopting appropriate regulations. Of the 27 coastal counties in Florida where sea turtles are known to nest, 22 have passed beachfront lighting ordinances in addition to 58 municipalities (http://www.myfwc.com/media/418420/seaturtle_lightordmap.pdf). Local governments have realized that adopting a lighting ordinance is the most effective method to address artificial lighting along the beachfront. While a majority of coastal local governments and counties have adopted beachfront lighting ordinances, compliance and enforcement is lacking in some areas. Further, the lighting in areas outside the beachfront ordinance coverage areas continues to be unregulated resulting in urban glow. Even protected beaches where light pollutions is minimized are subject to surrounding sky glow. During the 2010-2011 sea turtle nesting season in Florida alone, over 2,810 hatchlings were documented reported toFWC as being disoriented, 2,110 green turtle hatchlings were documented reported to FWC as being disoriented, a portion of which were green turtles, and 17 nesting greens were disoriented due to artificial lighting, and 7 nesting green turtles were disoriented in 2011 (KR SchanzleTrindell, Florida Fish and Wildlife Commission, pers. comm, 2011-2014). In addition, many hatchling disorientations likely are unreported.

Non-native vegetation has invaded many coastal areas and often outcompetes native plant species. Exotic vegetation may form impenetrable root mats that can invade and desiccate eggs, as well as trap hatchlings. The Australian pine (Casuarina equisetifolia) is particularly harmful to sea turtles. Dense stands have taken over many coastal areas throughout central and south Florida. Australian pines cause excessive shading of the beach that would not otherwise occur. Studies in Florida suggest that nests laid in shaded areas are subjected to lower incubation temperatures, which may alter the natural hatching sex ratio (Marcus and Maley, 1987; Schmelz and Mezich, 1988). Fallen Australian pines limit access to suitable nest sites and can entrap
nests females (Reardon and Mansfield, 1997). The shallow root network of these pines can interfere with nest construction (Schmelz and Mezich, 1988). Davis and Whiting (1977) reported that nesting activity declined in Everglades National Park where dense stands of Australian pine took over native dune vegetation on a remote nesting beach. Sisal, or century plant, (*Agave americana*) is native to arid regions of Mexico. The plant was widely grown in sandy soils around Florida in order to provide fiber for cordage. It has escaped cultivation in Florida and has been purposely planted on dunes. Although the effects of sisal on sea turtle nesting are uncertain, thickets with impenetrable sharp spines are occasionally found on developed beaches.

The countries in the Greater Antilles include Cuba, the Cayman Islands, Jamaica, Haiti, Dominican Republic, and Puerto Rico (affiliated with the U.S). There are protected beaches on the Guanahacabibes National Park in Cuba. Nests laid on these beaches and other minor nesting beaches are often completely destroyed during tropical storm events as was the case from 2001 and 2003 (Ricardo *et al.*, 2006). There are less than 25 green turtle nests laid in the Cayman Islands. These nests are also regularly eroded. Nesting habitat is threatened by vehicle use, artificial lighting, and associated recreational activities such as beach cleaning and beach equipment (Dow *et al.*, 2007). Very few green turtle nests are laid in Jamaica, where nesting beaches are under such substantial threat from development, disturbance, and pollution that nesting no longer occurs on many beaches (Haynes-Sutton *et al.*, 2011). In addition, a recent survey reported that more than half of the Jamaican beaches showed signs of oil pollution (Haynes-Sutton *et al.*, 2011). In Haiti, green turtles nest in much smaller numbers than they did historically, and nests are susceptible to coastal development from tourism and sand removal from the beaches (Ottenwalder, 1987). The beaches of the Dominican Republic host up to 100 green turtle nests per year on two beaches (Nagua and Boca del Estero), and are threatened by tourist related activities such as regular beach cleaning, vehicle use, and recreational equipment on the beach (Dow *et al.*, 2007).

Smaller nesting sites in Belize and Jamaica continue to be affected by activities associated with tourism such as coastal development and beachfront lighting (Smith *et al.*, 1992; Haynes-Sutton *et al.*, 2011). Indeed, in Belize there is an increasing threat to nesting beaches as a result of the growing tourist industry. Beachfront development has an added effect of increasing the artificial lighting, human activity, and pollution associated with sewage and waste disposal to once isolated nesting grounds. Debris is a serious problem on some beaches (Smith *et al.*, 1992; Dow *et al.*, 2007).

In the Bahamas, there are fewer than 100 nests a year. These nesting beaches face erosion from increased storm events and high tides. Sand mining causes increased erosion on the Bahamas (K. Bjorndal and A. Bolten, University of Florida, pers. comm., as cited in Dow *et al.*, 2007).

**Neritic/Oceanic Zones**

The Atlantic waters around Florida include important foraging habitats and migration corridors that support green turtles from hatchlings to adults (Bovery and Wynenken, 2013). Each life stage in these waters is affected by the degradation of these habitats.
Green turtles in the post-hatchling and early-juvenile stages are closely associated with Sargassum algae in the Atlantic and Gulf of Mexico (Witherington et al., 2012). Sargassum aggregates in convergent zones where pollutants such as tar balls and plastics also accumulate. Due to their size, turtles in these stages are more vulnerable as a result of ingesting these contaminants (Witherington, 2002).

Juvenile and adult green turtles forage in the neritic waters of coastal lagoons and along nearshore reefs. Coastal lagoons in Florida such as the Indian River Lagoon expose green turtles to high levels of pollutants as a result of agricultural and residential pollutants runoff (Hirama and Ehrhart, 2007). Increased nutrient load in these coastal waters causes eutrophication. Eutrophication, which is linked to harmful algal blooms, results in the loss of seagrass beds (Milton and Lutz, 2003). Green turtles have a higher incidence of FP in these waters compared to other nearby habitats such as the nearshore reef (Borrowman, 2008). The susceptibility to disease from this exposure is discussed in Section 5.2.5.3.

Boat propeller scarring further degrades seagrass beds. Scarred seagrass beds have been observed in all areas throughout the coastal waters of Florida. The most severe scarring occurs in areas where green turtles are known to foraging such as the Florida Keys and north Indian River Lagoon (Sargent et al., 1995).

Sand placement projects along the Florida coastline impact nearshore reefs as a result of direct burial of portions of the reef habitat and loss of food sources available to green turtles (Lindeman and Snyder, 1999).

Periodic dredging of sediments from navigational channels is carried out at large ports to provide for the passage of large commercial and military vessels. In addition, sand mining (dredging) for beach renourishment and construction projects occurs in the North Atlantic along the U.S., Mexico, and Central American coasts. Channelization of inshore and nearshore habitat and the subsequent disposal of dredged material in the marine environment also destroys or disrupts resting and foraging grounds (including grass beds and coral reefs) and affects nesting distribution by altering physical features in the marine environment (Hopkins and Murphy, 1980).

Conception Island Creek in the Bahamas supports a population of immature green turtles. Conception Island is an uninhabited wildlife sanctuary (Bjorndal and Bolten, 1996). Anchor damage is a threat to seagrass throughout this DPS. In the Bahamas and Panama, damage to seagrass as a result of anchor damage has been reported along with propeller scarring, dredging, sand mining, and marina construction. All degrade the seagrass habitat although the extent of this damage is not known (Dow et al., 2007). Tortuga Bay in Puerto Rico is used frequently by recreational boaters and their anchors have destroyed seagrass beds used for foraging by green turtles (Patrício et al., 2011). Seagrass beds in Jamaica and the Dominican Republic are often uprooted during dredging for coastal development projects such as dredging and the construction of new marinas. Fish kills and harmful algal blooms in Kingston Harbour and along the...
northeastern and southeastern coasts are indicative of serious problems in the coastal marine environment (Dow et al., 2007; Haynes-Sutton et al., 2011).

The Robinson Point area has the largest area of dense seagrass inside the Belize Barrier Reef (Searle, 2003). In Belize, erosion and runoff from agricultural activities increases the sediment load in nearshore waters, which can reduce the productivity of seagrass meadows (Smith et al., 1992). Also in Belize, increased boat traffic and channeling to allow for boat traffic has impacted sea-grasses, resulting in reduced foraging areas for green turtles (Smith et al., 1992).

In Panama, seagrass beds are degraded as a result of direct damage by divers and agricultural and residential runoff (Meylan et al., 2013).

In Haiti, green turtle foraging habitat is degraded due to the pollution by sewage runoff and marine debris near developed area degrading the seagrasses (Dow et al., 2007).

- Green turtle foraging in the waters surrounding Bermuda are threatened with the net loss and degradation seagrass beds from 1997 to 2004, (Murdoch et al., 2007).

5.2.5.2. Factor B: Overutilization

The harvesting of eggs and turtles was likely a factor that contributed to the historical declines of the population. Current legal and illegal harvest of green turtles continues in the eastern Atlantic and the Caribbean for human consumption.

Egg and Turtle Harvest

The present distribution of the breeding sites has been largely affected by historical patterns of human exploitation. The only substantial breeding colonies left today are those that have not been permanently inhabited by humans or have not been heavily exploited until recently (Groombridge and Luxmoore, 1989). This demographic trend is corroborated by the fact that several islands which formerly held large breeding colonies are known to have lost them once becoming inhabited by humans (e.g., Bermuda; King, 1982). The Cayman Island nesting site, formerly one of the largest green turtle rookeries, has been largely affected by historical patterns of human exploitation.

A partial list of the countries within the North Atlantic DPS where ongoing intentional capture of green turtles occurs, includes Costa Rica (Mangel and Troeng, 2001; Mangel et al., 2001), Mexico (Seminoff, 2000; Gardner and Nichols, 2001), Cuba (Fleming, 2001), Nicaragua (Lagueux, 1998), the Bahamas (Fleming, 2001), and the Cayman Islands (Fleming, 2001). Despite substantial declines in green turtle population sizes, harvest remains legal in several of these countries [Humphrey and Salm, 1996; Wamukoya et al., 1996; Fleming, 2001; Fretney, 2001; Bräutigam and Eckert, 2006].

In the 1970s, adult females were harvested extensively on the nesting beach in Tortuguero, Costa Rica. A mean of 600 adults were killed annually from 1997 to 1999 with a peak of 1,720 nesting adults poached in 1997 (Troeng, 1998; Troeng and Rankin, 2005). Within this timeframe,
despite beach protection efforts, a mean of 9.8 percent of nests in Tortuguero were poached for eggs (Troëng, 2000a). Today, due to conservation efforts, the poaching of nests and females has been reduced. More recent harvest of nests and nesting adults continues at Tortuguero. In 2007, 183 green turtle nests and 19 nesting females were poached from the northern section of Tortuguero Beach. In 2011, 181 green turtle nests (1.5 percent of the total number of nests documented) were recorded as poached during daily nesting surveys (Prieto and Harrison, 2012). In 2012, 32 adult green turtles were documented as being poached on the Tortuguero beach (Prieto and Harrison, 2012). The nesting colony at Tortuguero has exhibited encouraging trends since the early 1990s (Bjorndal et al., 1999; Prieto and Harrison, 2012).

In the Yucatan Peninsula, less than 30 years ago, sea turtles were over-exploited and numbers diminished. In 1973, a Mexican law provided complete protection to sea turtles within the Mexican Gulf of Mexico (Duran Najera, 1990). At Rio Lagartos and on Isla Holbox in the Yucatan Peninsula in Mexico, the main threat continues to be egg harvesting (Duran Najera, 1990). On Aguada beach, adult harvesting continues to be a major threat (Duran Najera, 1990; Guzmán-Hernández and García Alvarado, 2011). On the beaches of Alto Lucero in Vera Cruz, Mexico, nesting turtles are commonly slaughtered (Dirado et al., 2002).

For many years, there was a directed commercial fishery on sea turtles in Cuba. In 2008, that fishery was closed. The fishery had operated since 1968 and can be broken down into four periods. The first period was from 1968 to 1975, when there were no regulations. The second period was from 1976 to 1987, when there was a closed season from June through August. During this second period about 3,200 sea turtles per year were taken. The third period was from 1988 to 1994, during which the closed season was expanded to be implemented from May through August given the importance of May for reproduction. This period was also characterized by serious economic difficulties in Cuba, which resulted in higher fuel costs and less fishing effort. It was estimated that only 300 individuals per year were taken. The fourth period occurred when the fishery was constrained to two sites, with a maximum quota of 25 tons for greens and loggerheads. The principle use of turtle meat was for food and products (F. Moncada Gavilán, Ministerio de la Industria Pesquera, pers. comm., 2013). Although the number of fishing boats operating in Cuba waters decreased between 1979 and 1996, the catch per unit effort increased (Blanco and Cardona, 1983 as cited in Gavilan and Andreu, 1998). The green turtle fishery is now closed, but turtles taken as bycatch in other fisheries can still be used in two communities; elsewhere they cannot be utilized and must be discarded (Gavilan et al., 2003).

The greatest current threats to green turtles in Cuba are illegal or stealth fishing of sea turtles, as well as bycatch. In 2008, about 10,000 kg of sea turtle meat was confiscated. The punishment for this crime is 1,000-5,000 pesos, which is high for Cubans. Cuba is contemplating increasing the fine. The Office of National Fishing Inspection is also increasing efforts to address this illegal fishing (F. Monrado, Ministerio de la Industria Pesquera, pers. comm., 2013).

In Panama, green turtles continue to be harvested albeit at lower levels than 30 years ago where more than 300 adult green turtles and hawksbills were captured by nets within 3 months. Mating pairs of green turtles continue to be captured by harpoons and nets in areas outside of the town of
Bocas del Toro. Green turtles are also captured by lobster divers opportunistically (Meylan et al., 2013).

In Nicaragua, green turtles have been legally harvested for more than 400 years. From 1967 to 1977 green turtles were harvested for local and foreign consumption, including annual exports to the U.S. and Europe in excess of 10,000 turtles. Processing plants have been closed for over 30 years—after Nicaragua became a signatory of CITES in 1977. The primary foraging area for Nicaragua is reported to have large juvenile and adult survivorship at 0.55, likely due to the ongoing directed take of green turtles in this area (Campbell and Lagueux, 2005). The commercial artisanal green turtle fishery in Nicaragua continues to threaten the largest remaining green turtle population in the Atlantic (Campbell and Lagueux, 2005). Local demand for turtle meat in coastal communities has continued (Garland and Carthy, 2010). This is a legal turtle fishery on the Caribbean coast and is in the most important developmental and foraging habitat for Caribbean green turtles (Fleming, 2001), including those nesting in the Bahamas, Bermuda, Florida and, importantly, Tortuguero, Costa Rica (Campbell and Lagueux, 2005). In the Miskito Cays along the Caribbean coast of Nicaragua, an area considered to be the primary foraging habitat for turtles originating from Tortuguero, a mean of 9,357 turtles were killed annually between 1994 and 1996 (Lagueux, 1998). Despite hunting in the Caribbean waters of Nicaragua, Columbia, Honduras, Panama, and Venezuela, of juveniles to adults, the numbers of nests nesting colony at Tortuguero continues to increase was not depleted possibly due to the variety of size classes that were harvested (SWOT, 2011).

Prior to 2009, when the government declared a complete ban on the harvesting of sea turtles, sea turtle exploitations were still legal in the Bahamas (Bjorndal and Bolten, 2009). Union Creek Marine Reserve is in a national park of the Bahamas National Trust and serves as foraging area for immature green turtles (Bjorndal, 2008) which has been protected from exploitation over the last three decades. Bjorndal et al. (2003) found that after green turtles left Union Creek, the annual survival probability declined since they were no longer protected from human-induced mortality.

The Cayman Island nesting site was historically one of the largest rookeries in the world. Nesting was nearly eliminated with the onset of the turtle fishery (Lewis, 1940; Parsons, 1962). Within the last decade, green turtles continue to nest at low levels (Aiken et al., 2001). These nests may be a result of re-colonization by turtles from nearby nesting sites (Wood and Wood, 1994). In 1998, nest monitoring began by the Cayman Islands Department of Environment’s Marine Turtle Beach Monitoring Program. Seven years of intensive monitoring showed green turtles still nest in low numbers on the islands. The low nesting numbers coupled with the present threats, such as legal harvesting, call into question the viability of this population over the long-term (Bell and Austin, 2003; Solomon et al., 2006). Sea turtles can still be caught legally during the open season (1 November through 30 April). Each licensed fisherman (approximately 25) can take no more than six turtles per season and each must weigh more than 80 pounds (Cayman Islands Government, 1996 as cited in Aiken et al., 2001).

In Jamaica, in the 1850s, green turtles were the most common sea turtle species but, by the 1940s nests were rare (Lewis, 1940). Green turtles were used for local consumption as well as shipped to England. While the decline in Jamaican populations of green turtles occurred many years ago,
the continued harvesting of green turtles in their foraging grounds make reestablishment of a viable population in Jamaica difficult (Haynes-Sutton et al., 2011).

In Puerto Rico, despite turtles and egg harvesting being outlawed in the 1970s, there continues to be demand for sea turtle meat, eggs, and products (Fleming, 2001). A conservation project was initiated in Vieques in 1991. Since that time, the harvesting of eggs and nesting females have been controlled (Fleming, 2001).

The overexploitation of green turtles foraging in coastal waters of Panama is the principal cause of population depletion (Peckham et al., 2007; Ruiz et al., 2007). Direct take of eggs is also an ongoing problem in Panama (Evans and Vargas, 1998).

During the period from 1650 to 1900, green turtles were actively hunted in the waters of Belize. In decades past, hundreds of green turtles were reported to have been nesting on Northern Two Cays (Sandbore and Northern Cay). Smith et al. (1992) reported that in the 1860s, approximately 2,000-6,000 live turtles were exported annually. Searle et al. (2001) reported 83 green turtles were in the Belize City markets, which represents approximately 25 percent of the total captures for markets throughout Belize, estimating more than 300 green turtles in one year were harvested and sold in the markets. Green turtles nest occasionally in Belize but not in the numbers that were once recorded (Smith et al., 1992). Green turtles nesting on Belize’s beaches and foraging along its coast are harvested in the Robinson Point area and sold in markets and restaurants (Searle, 2003). Large numbers of green turtles are captured in the area southeast of Belize which may be an important migratory corridor (Searle, 2004).

In the 1970s, there was a Florida green turtle fishery that harvested a large number of green turtles from Florida waters (Parsons, 1962; Witzell, 1994). While these threats have been largely eliminated in Florida due to successful conservation measures, the hunting of juvenile and adult turtles continues both legally and illegally in many foraging areas where green turtles originating from Florida are known to occur (Fleming, 2001; Chacón, 2002). Although there has been a steady increase in nesting numbers in Florida since index nesting beach surveys began in 1989, current nesting activity likely represents only a fraction of historical levels (NMFS and U.S. FWS, 2007)(NMFS and USFWS, 2007).

Nesting green turtles were extirpated in Bermuda due to long term overexploitation. Despite an attempt in 1959 to reestablish the nesting population, there are no recorded green turtles nesting in Bermuda (Schoch et al., 2006).

Green turtles have been observed nesting on National Park Banc d’Arguin, Mauritania (Fretey, 2001). Although the frequency of green turtle nesting in Mauritania is not known, green turtle nests are reported as being harvested there (Fretey and Hama, 2012). There are limited data on green turtle harvesting in Mauritania. In addition, Banc d’Arguin supports one of the most important foraging areas for green turtles in the Atlantic coast of Africa (Fretey, 2001). Fishermen have long been documented catching turtles with nets and harpoons for consumption (Fretey and Hama, 2012). Direct capture of nesting turtles for consumption by residents also occurs (Flores et al., 2006).
Factor C: Disease or Predation

Disease (especially FP) and predation are continuing threats to the North Atlantic DPS. Harmful algal blooms also affect turtles in the marine environment.

Epidemiological studies indicate increased incidence of this disease in portions of this DPS. The extent to which this will affect the long-term outlook for green turtles in the North Atlantic DPS is unknown but is of concern.

FP has been found in green turtle populations of the Bahamas, the Dominican Republic, Puerto Rico (Dow et al., 2007; Patricio et al., 2011), Cayman Islands (Wood and Wood, 1994; Dow et al., 2007), Costa Rica (Fortuguero; Mangel and Troëng, 2001; Mangel et al., 2001), Cuba (Moncada and Prieto, 2000), Mexico (Yucatan Peninsula; K. Lopez, pers. comm., as cited in MTSG, 2004), Nicaragua (Lagueux, 1998), and the United States (Florida; Hirama and Ehrhart, 2007; Hirama, 2001; Foley et al., 2005; Enel, 2005).

FP was first documented in Florida in 1938 and has been persistent in the Indian River Lagoon population for 30+ years at varying levels of prevalence, (28–72 percent; Hirama and Ehrhart, 2007; Schroeder et al., 1998). In Florida, 22 percent of the 6,027 green turtles stranded from 1980–2005 had external fibropapilloma tumors, suggesting serious consequences for population stability (Singel et al., 2003; FWC, 2007 as cited in NMFS and USFWS, 2007). FP continues to be a major problem in the Indian River Lagoon system and along the nearshore reefs of central eastern Florida (8–21 percent in 1989 to 1996). A correlation exists between these degraded habitats and the prevalence of FP in the green turtles that forage in these areas (Aguirre and Lutz, 2004; Foley et al., 2005). Herbst and Klein (1995) cautioned that although field observations indicate an association between FP and degraded habitats types and potential contaminants, no direct link has been established, and virulent pathogens (if that is indeed the ultimate cause of the disease) do not necessarily require an immunosuppressed host.

Response: Added in text.

Comment [A54]: Makes no mention of FP’s correlation with degraded environments. Aguirre and Lutz 2004, Foley et al. 2005 (already cited), HOWEVER: Eff ect of FP on population dynamics is not well defined, qualitatively or quantitatively (Herbst & Klein 1995)

No conclusive estimates of the effect of FP on mortality or reproduction exist (Chaloupka & Balazs 2005)

Herbst and Klein (1995) caution that although field observations indicate an association between FP and degraded habitats types and potential contaminants, no direct link has been established, and virulent pathogens (if that is indeed the ultimate cause of the disease) do not necessarily require an immunosuppressed host.

Response: Added in text.


FP was first documented in Port Canaveral, a site relatively close to these aforementioned sites (Hirama and Ehrhart, 2007). A comparison of FP rates at two sites near St. Lucie County, Florida, found a similar pattern of presence/absence, with the intake canal of the St. Lucie Nuclear Power Plant showing low incidence (2.3 and 12.6 percent in 2008 and 2012; M. Bresette, Inwater Research Group, pers. comm., 2013) vs. high incidence in the nearby Indian River Lagoon (59.4 and 70.2 percent in 1999 and 2000, respectively; Bresette et al., 2005). Between 1980 and 1998, all green turtle strandings with signs of fibropapilloma tumors were found in southern Florida where over 20 percent of all green turtles exhibited the disease (Foley et al., 2005). Since 1998, some green turtles with fibropapillomas have stranded in northeast and northwest Florida (A. Foley, Florida Fish and Wildlife Conservation Commission, pers. comm., 2007). In addition, about 7 percent (109 out of 1648) of the green turtles found cold-stunned in St. Joseph Bay, Florida, during a mass cold stunning event in January 2010 had fibropapilloma-like tumors. However, only 5 percent (two out of 388) green turtles found cold-stunned in this area had tumors (Foley et al., in press). The qualitative and quantitative effect of FP on the green turtle population has not established (Herbst and Klein, 1995). Despite the high incidence of FP among foraging populations, there is no conclusive evidence on the effect of FP on reproductive effort (Chaloupka and Balaz, 2005). Green turtles in Florida have demonstrated encouraging signs of recovery after more than 20 years of protection efforts with the population increasing at a rate of 13.9 percent per year (Chaloupka et al., 2008b).
Prior to 2010, there were no confirmed reports of the external fibropapilloma tumors in green turtles found in Texas waters. The first two were confirmed in June 2010. Subsequent entanglement net captures found more than a 33 percent infection rate for green turtles in developmental foraging grounds in lower Laguna Madre, Texas (Prieto et al., 2012).

Elsewhere in the North Atlantic, external fibropapilloma tumors have been seen in green turtles in the Belize City market. Some turtle fishermen report catching green turtles with fibropapillomas (locally called warts) so abundant that they would not sell or eat the turtle (Smith et al., 1992).

On Isla Contoy in the Yucatan Peninsula, Mexico, nesting green turtles were found to be infected by parasitic leech (Ozobranchus branchiatus) attached predominantly around the neck area. These turtles were thought to exhibit signs of FP (Duran Najera, 1990).

Although this disease is of major concern in some green turtle populations, it should be noted that photographic evidence from Hawaii and Florida shows that the tumors on some green turtles go into regression (Hirama, 2001; Hirama and Ehrhart, 2007) and in some cases the presence of FP may not hinder an individual’s growth (Chaloupka and Balazs, 2005). The implications of these studies are still not fully understood, although it is indicative that FP is not always lethal.

Harmful algal blooms, such as a red tide, also affect green turtles in the North Atlantic DPS. In Florida, the species that causes most red tides is Karenia brevis, a dinoflagellate that produces a toxin (Redlow et al., 2002). Since 2007, there were two red tide events, one in 2007 along the east coast of Florida, and one in 2012 along the west coast of Florida. Sea turtle stranding trends indicated that these events were acting as a mortality factor (A. Foley, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013). Sea turtles that washed ashore alive during these red tide events displayed symptoms that were consistent with acute brevitoxicosis (i.e., uncoordinated and lethargic behavior but otherwise robust and healthy in appearance) and completely recovered within days of being removed from the area of the red tide. The population level effects of these events are not yet known.

With respect to predation, predation of nests and hatchlings is a continuing threat and, in the absence of well managed nest protection programs, predators may take significant numbers of eggs; however, nest protection programs are in place at most of the major nesting beaches in the North Atlantic DPS. Depredation rates from some species such as raccoons (Procyon lotor), feral hogs (Sus scrofa), foxes (Urocyon cinereoargenteus and Vulpes vulpes), and coyotes (Canis latrans), are managed at various levels and degrees of effectiveness throughout the DPS. Those species that are difficult to impossible to manage include red fire ants (Solenopsis invicta) and jaguars (Panthera onca).

The most common predators at the primary nesting beaches in the southeastern United States are ghost crabs (Ocypode quadrata), raccoons (Procyon lotor), feral hogs (Sus scrofa), foxes (Urocyon cinereoargenteus and Vulpes vulpes), coyotes (Canis latrans), armadillos (Dasypus novemcinctus), and red fire ants (Solenopsis invicta) (Stancyk, 1982). On At Tortuguero beach...
National Park, green turtles killed by jaguars have increased since 1997, with 22-57 killed in 1999-2011 (Troëng, 2000b; Prieto and Harrison, 2012).

Fire ants have been shown to cause high hatchling mortality in Florida (Allen et al., 2001). In Tortugero, Costa Rica, ants were noted in green turtle nests (Fowler, 1979; Mangel and Troëng, 2001; Fowler, 1979; Mangel et al., 2001). Fowler (1979) found that ants invaded 35 of 237 (14.8%) green turtle nests where they fed on hatchlings and eggs, although it was unclear whether they were feeding on dead or weak hatchlings (Fowler, 1979). Mangel et al. (2001) noted ants depredated unhatched eggs, pipped eggs and hatchlings. Wetterer (2006) found that fire ants were by far the most common ant at Tortuguero, Costa Rica. The presence of vertebrate predators such as dogs and raccoons also affect hatchlings as well as unhatched eggs (Engeman et al., 2005). While these threats have been mitigated in some areas such as Florida (Engeman et al., 2005), they are very problematic in other areas and have led to catastrophic egg and hatchling mortality in some cases.

Predation of eggs and hatchlings by native and introduced species occurs on almost all nesting beaches throughout the North Atlantic DPS. On Cuba, sea turtle nests are depredated by feral pigs and dogs although the depredation on green turtle nests specifically is not known (Dow et al., 2007).

Eggs and hatchlings on Isla Contoy and Rio Lagartos, Yucatan Peninsula, Mexico, are heavily predated by ants, raccoons, foxes, and feral dogs (Duran Najera, 1990; Zambrano and Rodriguez, 1995).

Green turtles are heavily affected by sharks in some areas such as Puerto Rico and Panama (Dow et al., 2007), but terrestrial predators such as ants and terrestrial vertebrates appear to be a much larger problem for green turtle survival.

Harmful algal blooms, such as a red tide, also affect green turtles in the North Atlantic DPS. In Florida, the species that causes most red tides is *Karenia brevis*, a dinoflagellate that produces a toxin (Redlow et al., 2002). Since 2007, there were two red tide events, one in 2007 along the east coast of Florida, and one in 2012 along the west coast of Florida. Sea turtle stranding trends indicated that these events were acting as a mortality factor (A. Foley, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013). Sea turtles that washed ashore alive during these red tide events displayed symptoms that were consistent with acute brevitoxicosis (i.e., uncoordinated and lethargic behavior but otherwise robust and healthy in appearance) and completely recovered within days of being removed from the area of the red tide. The population level effects of these events are not yet known.

5.2.5.4. Factor D: Inadequacy of Existing Regulatory Mechanisms

Regulatory mechanisms are in place in many areas that should address direct and incidental take of North Atlantic DPS green turtles; however, these regulatory mechanisms are not in place throughout the DPS, and some are insufficient or are not being implemented effectively. The inadequacy of existing regulatory mechanisms for impacts to nesting beach habitat and overutilization (harvest of turtles and eggs) are continued threats to this DPS (see Factors A and B, above). In addition, in the following section (Factor E), we describe the insufficiency of
regulatory mechanisms in relation to several threats including incidental bycatch in fishing gear, boat strikes, port dredging, debris, national defense, toxic compounds, and climate change. Despite the existing regulatory mechanisms, threats to nesting beaches, eggs, hatchlings, juveniles, and adults through habitat degradation, harvest, and incidental harm occur throughout the North Atlantic DPS.

In addition to local and national regulatory mechanisms, there are a minimum of ten national and international treaties and/or regulatory mechanisms that pertain to the North Atlantic DPS. Hykle (2002) and Tiwari (2002) reviewed the value of some international instruments, which vary in their effectiveness. Often, international treaties 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, 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 a 2002 special issue of the Journal of International Wildlife Law and Policy: International Instruments and Marine Turtle Conservation (Hykle, 2002).

5.2.5.5. Factor E: Other Natural or Manmade Factors

Fishery bycatch that occurs throughout the North Atlantic DPS, particularly trawling, gill net, and dredging, is a continued threat to this DPS. Additional threats from interactions with different types of fishing gear, boat strikes, climate change and natural disasters negatively affect this DPS

5.2.5.5.1. Incidental Bycatch in Fishing Gear

Fisheries bycatch in artisanal and industrial fishing gear is a major threat to green turtles in the North Atlantic DPS. Although other species such as leatherback and loggerhead turtles have received most of the attention relative to sea turtle bycatch, green turtles are also susceptible, particularly in nearshore artisanal fisheries. These gear include drift nets, set nets, pound nets, and trawls. Their adverse impacts on sea turtles have been documented in marine environments throughout the world (National Research Council, 1990b; Epperly, 2003; Lutcavage et al., 1997). The lack of comprehensive and effective monitoring and bycatch reduction efforts in many pelagic and near-shore fisheries operations throughout the North Atlantic DPS still allows substantial direct and indirect mortality (NMFS and FWS, 2007).

Gill net and Trawl Fisheries

Gill net fisheries may be the most ubiquitous of fisheries operating in the neritic range of the North Atlantic DPS. Murray (2009) conducted a comprehensive examination of sea turtle bycatch by sink gillnet gear in the U.S. mid-Atlantic region reporting low numbers of green turtles caught incidentally. Comprehensive estimates of bycatch in gill net fisheries do not exist. In the U.S., some states (e.g., South Carolina, Georgia, Florida, Louisiana, and Texas) have prohibited gill nets in their waters, but there remain active gill net fisheries in other U.S. states, in U.S. federal waters, Mexican waters, Central and South America, and the Northeast Atlantic.

Comment [A58]: Makes no mention of trawl captures in U.S. Perhaps change header to: fisheries?? Since more than these fisheries are mentioned/addressed.
Response: Revised and added in text.

Comment [A59]: See: Murray 2009 before stating this re: gillnets.
Pound nets are fixed gear composed of a series of poles driven into the bottom upon which netting is suspended. Pound nets basically operate like a live-trap with the pound constructed of a series of funnels leading to a bag that is open at the top, and a long leader of netting that extends from shallow to deeper water where the pound is located. In some configurations, the leader is suspended from the surface by a series of stringers or vertical lines. Sea turtles incidentally captured in the open top pound, which is composed of small mesh webbing, are usually safe from injury and may be released easily when the fishermen pull the nets. However, sea turtle mortalities have been documented in the leader of certain pound nets and have been recorded in North Carolina. Epperly et al. (2007) recorded 246 green turtles incidentally captured in the North Carolina pound net fishery between 1995 and 2001. Large mesh leaders (greater than 12-inch stretched mesh) may act as a gill net, entangling sea turtles by the head or foreflippers [Bellmund et al., 1987; Mansfield 2006]. In 2002, the U.S. prohibited, in certain areas within the Chesapeake Bay, with rare occurrences of green turtles, and at certain times, pound net leaders having mesh greater than or equal to 12 inches and leaders with stringers (67 FR 41196, June 17, 2002); although, green turtle mortalities have been recorded in the Chesapeake Bay are rare. Subsequent regulations have further restricted the use of certain pound net leaders in certain geographic areas and established pound net leader gear modifications (69 FR 24997, May 5, 2004; 71 FR 36024, June 23, 2006).

While a directed turtle fishery is no longer a threat in Cuba, bycatch remains a threat. Finfish fisheries accounted for the greatest proportion of turtle bycatch (53 percent). The highest incidence of capture was in trawl nets, which are used in various habitats, but generally in shallow areas. Cuba enacted resolution 58/2004 to reduce this type of fishing.

In Jamaica, fish traps and gill nets are the gear primarily identified in sea turtle bycatch. These gear types are used predominantly in Jamaican waters (Bjorkland et al., 2008). Purse seine and gill nets are used commonly in the waters of the Dominican Republic. Bycatch estimates are not available but this type of fishing gear has been known to cause the mortality of green turtles (Dow et al., 2007).

In Costa Rica, gill nets, hook and line, and trawls are the main gear types deployed. (Food and Agriculture Organization of the United Nations, 2004). No sea turtle bycatch data is available. Shark-netting operations in Panama are known to capture green turtles (Meylan et al., 2013).

The development and implementation of Turtle Excluder Devices (TEDs) in the shrimp trawl fishery was likely the most significant conservation accomplishment for North Atlantic green turtles in the marine environment since their listing. In the southeast U.S. and Gulf of Mexico, TEDs have been mandatory in shrimp and flounder trawls for over a decade. However, TEDs are not required in all trawl fisheries, and green turtle mortality continues where shrimp trawling in the Gulf of Mexico is the highest source of sea turtle bycatch (Lewison et al., 2013). Based on 1997 to 1998 green turtle bycatch rates in the southeast U.S. and waters of the Gulf of Mexico, a 2002 study estimated 48,239 interactions with green turtles and shrimp trawls (Epperly et al., 2002). The estimated number of green turtles injured and/or killed by shrimp trawls each year...
In the Gulf and U.S. Southeast Atlantic combined is between 4,620 and 7,055 with the current regulations in place [NOAA., 2002].

In 1995, a high number of strandings occurred on the south coast of Guatemala due to drowning in fishing gear; mainly shrimp trawls during the beginning of the nesting season (June and July). Patrols began by the Guatemalan Navy, and the shrimp trawlers moved further offshore. There were no more strandings for the rest of the season. In 1996, the Guatemalan shrimp fleet installed Turtle Excluder Devices (TEDs), which reduced incidental capture and drowning of turtles (Juarez and Muccio, 1997). During a survey conducted in 2002 along two beaches on the Caribbean coast of Guatemala, stranded turtles were found mutilated, and injured from fishing nets—an indication of the impact of the commercial and local fisheries on sea turtles including the green turtle (Montes-Osorio et al., 2007).

Bycatch data reported from Mauritania include 17 green turtles (16 females and 1 male) weighing 40–50 kg in an artisanal purse seine on 27 September 1980 at Marguerite Island (Maigret, 1983; Arvy et al., 1996).

**Dredging**

Dredge fishing gear is the predominant gear used to harvest sea scallops off the mid- and northeastern U.S. Atlantic coast. Sea scallop dredges are composed of a heavy steel frame and cutting bar located on the bottom part of the frame and a bag made of metal rings and mesh twine attached to the frame. Turtles can be struck and injured or killed by the dredge frame and/or captured in the bag where they may drown or be further injured or killed when the catch and heavy gear are dumped on the vessel deck. In addition to the destruction or degradation of habitat described in Section 5.2.6.1, periodic dredging of sediments from navigational channels can also result in incidental mortality of sea turtles. Direct injury or mortality of green turtles by dredges has been well documented in the southeastern and mid-Atlantic U.S. (National Research Council, 1990b). From 1980 to 2013, 105 green turtles were impacted as a result of dredging operations in the U.S Atlantic and Gulf of Mexico. Solutions, including modification of dredges, have been successfully implemented to reduce mortalities and injuries in the United States (Nelson and Shafer, 1996; NMFS, 1991), and annual take limits are imposed by NMFS based on the expected number of green turtles impacted that will not directly or indirectly, appreciably reduce in the likelihood of survival and recovery of the green turtle in the wild.

**5.2.5.5.2. Vessel Strikes and Boat Traffic**

Boat strikes have been shown to be a major mortality source in Florida (Singel et al., 2003). It is quite likely that this is a chronic, albeit unreported, problem near developed coastlines in other areas as well, such as Panama (e.g., Orós et al., 2005). From 2005 to 2009, 18.2 percent of all stranded green turtles (695 of 3818) in the U.S. Atlantic (Northeast, Southeast, and Gulf of Mexico) were documented as having sustained some type of propeller or collision injuries. It is not known what proportion of these injuries was post- or ante-mortem (L. Belskis, NMFS, pers. comm., 2013).

Boat traffic has been shown to exclude green turtles from preferred coastal foraging pastures (Seminoff et al., 2002c), which may negatively affect their nutritional intake.
5.2.5.5.3. **Climate Change**

Similar to other areas of the world, climate change and sea level rise have the potential to impact green turtles in the North Atlantic. Weishampel *et al.* (2010) recorded median nesting date shifts earlier (~4.5 d per degree C) with higher May sea surface temperatures for Florida green turtles. The impact of this temperature shift is not known. Over the long term, North Atlantic turtle populations could be threatened by the alteration of thermal sand characteristics of beaches (from global warming), resulting in the reduction or cessation of male hatching production (Hawkes *et al.*, 2009; Poloczanska *et al.*, 2009). Further, a significant rise in sea level would restrict green turtle nesting habitat in the North Atlantic.

5.2.5.5.4. **Natural Disasters**

Another natural factor that has the potential to affect the recovery of green turtles is periodic hurricanes and other weather events. In general, these events are episodic and, although they may affect green turtle hatching production, the results are generally localized and they rarely result in whole-scale losses over multiple nesting seasons. The negative effects of hurricanes on low-lying and/or developed shorelines may be longer-lasting and a greater threat overall. Also, when combined with the effects of sea level rise, there may be increased cumulative impacts from future storms.

Cold stunning of green turtles regularly occur at several locations in the United States, including Cape Cod Bay, Massachusetts (Still *et al.*, 2002); Long Island Sound, New York (Meylan and Sadove, 1986; Morreale *et al.*, 1992); the Indian River Lagoon system and the panhandle of Florida (Mendonça and Ehrlhart, 1982; Witherington and Ehrlhart, 1989, Foley *et al.* 2007); and Texas inshore waters (Hildebrand, 1982; Shaver, 1990). In January 2010, a massive sea turtle cold stunning event occurred throughout the State of Florida. Although stranded turtles were rescued throughout the state, the two major epicenters of cold-stunning activities occurred in the vicinity of Cape Canaveral in Brevard County and St. Joseph Bay in Gulf County. An unusually prolonged period of very cold weather resulted in 4,613 cold-stunned turtles documented (A. Foley, Florida Fish and Wildlife Conservation Commission, pers. comm., 2012 as cited in Avens *et al.*, 2012). The majority (4,366) of the affected turtles were green turtles. Of the 4,613 turtles collected, an estimated 910 turtles died as a result of cold stunning. Approximately 85 percent of the dead turtles were found dead; only a small number of turtles that were found alive died after rescue (A. Foley, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013). In December 2010 and January 2011, a large number of green turtles in Florida were again affected by cold weather, but to a much lesser extent than was observed during the January 2010 event. Over 700 green turtles cold-stunned in December 2010 and January 2011; a portion of these died and the remaining turtles were released back into the wild following rehabilitation (A. Foley, Florida Fish and Wildlife Conservation Commission, pers. comm., 2012). In Texas, 459 green turtles cold-stunned in 2010. In 2011, 1,517 green turtles were reported cold stunned during the month of February; a portion of these died and the remaining turtles were released back into the wild following rehabilitation (D. Shaver, National Park Service, pers. comm., 2012).

5.2.5.5.5. **Contaminants and Marine Debris**

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**Comment [A68]:** Climate change gets represented well in threats, including storm effects, inundation, erosion, habitat loss, lethal nest temps, and feminization. That’s two threats categories at least. The team might consider whether this effect received correct weight in the Delphi (voting) methods, given climate trends.

Response: Added a summary to
Several activities associated with offshore oil and gas production, including oil spills, water quality (operational discharge), seismic surveys, explosive platform removal, platform lighting, and noise from drillships and production activities, are known to impact sea turtles (Conant et al., 2009; Davis et al., 2000; National Research Council, 1996; Viada et al., 2008; G. Gitschlag, NMFS, pers. comm., 2007, as cited in Conant et al., 2009). As of 2010, there were more than 3,400 federally regulated offshore platforms in the Gulf of Mexico dedicated to natural gas and oil production (Shapiro et al., 2013). Additional state-regulated platforms are located in state waters (Texas and Louisiana). There are currently no active leases off the Atlantic coast.

Oil spills near nesting beaches, just prior to or during the nesting season, place nesting females, incubating egg clutches, and hatchlings at significant risk from direct exposure to contaminants (Fritts and McGehee, 1982; Lutcavage et al., 1997; Witherington, 1999), as well as negative impacts on nesting habitat. Annually about 1 percent of all sea turtle strandings along the U.S. east coast have been associated with oil, but higher rates of 3 to 6 percent have been observed in South Florida and Texas (Teas, 1994; Plotkin and Amos, 1990; Rabalais and Rabalais, 1980).

Oil cleanup activities can also be harmful. Earth-moving equipment can dissuade females from nesting and destroy nests, containment booms can entrap hatchlings, and lighting from nighttime activities can misorient turtles (Witherington, 1999).

The Deepwater Horizon (Mississippi Canyon 252) oil spill, which started April 20, 2010, discharged oil into the Gulf of Mexico through July 15, 2010. According to government estimates, between 379 and 757 million liters (100 and 200 million gallons) of oil were released into the Gulf of Mexico during this time. The U.S. Coast Guard estimates that more than 189 million liters (50 million gallons) of oil have been removed from the Gulf, or roughly a quarter of the spill amount. Additional impacts to natural resources may be attributed to the 7 million liters (1.84 million gallons) of dispersant that were applied to the spill. The U.S. Coast Guard, the States, and Responsible Parties that formed the Unified Area Command (with advice from Federal and State natural resource agencies) initiated protective measures and cleanup efforts by preparing contingency plans to deal with petroleum and other hazardous chemical spills for each State’s coastline. These plans identified sensitive habitats, including all federally listed species’ habitats, which received a higher priority for response actions and allowed for immediate habitat protective measures coinciding with cleanup activities. Throughout the Deepwater Horizon oil spill response, the U.S. Coast Guard was responsible for and continues to oversee implementation and documentation of avoidance and minimization measures to protect trust resources, including sea turtles. Though containment of the well was completed in September 2010, other countermeasures, cleanup, and waste disposal are continuing and, therefore, a detailed analysis of the success of the avoidance and minimization measures has not been conducted. In addition, Natural Resource Damage Assessment studies regarding potential effects to fish and wildlife resources are currently being conducted along the northern Gulf of Mexico coast. It is not yet clear what all of the immediate and long-term impacts of the Deepwater Horizon oil well blowout and uncontrolled release has had, and will have, on green turtles in the Gulf of Mexico. However, green turtles, from post-hatchling to adult can be found mostly in the pelagic waters of the Gulf of Mexico with some foraging in the neritic zone (Witherington et al., 2012) note that the Deepwater Horizon oil spill was particularly harmful to pelagic juvenile green turtles.
In Cuba, Jamaica, Puerto Rico, and Panama, water quality is also affected by sewage and industrial and agricultural runoff. The occurrence of disease such as FP in green turtles may be an indication of poor environmental health (Aguirre and Lutz, 2004). Marine debris, ship pollution, and sedimentation affect the water quality in the Cayman Islands and Panama (Dow et al., 2007). Pollution remains a major threat in the waters of Jamaica. Major sources of pollution are industrial and agricultural effluent, garbage dumps and solid waste, and household sewage (Greenway, 1977; Green and Webber, 2003).

Green turtles are affected by anthropogenic marine debris throughout the North Atlantic DPS. The gut contents of turtles were analyzed, along the south coast of Texas, between 1983 and 1995. Sea turtles with ingested debris such as fishing line, glass, and plastic represented 51.7 percent of all turtles analyzed, with green turtles representing one of the species most affected (Shaver and Plotkin, 1998). In coastal waters of Florida, both ingestion of plastics and entanglement with fishing gear pose a threat to green turtles (Bjorndal et al., 1994). Juvenile green turtles in pelagic waters are particularly susceptible to these impacts as they feed on Sargassum where there is a high occurrence of debris (Wabnitz and Nichols, 2010; Witherington et al., 2012).

During 1997-2009, 481 out of the 5,347 stranded green turtles were reported as affected as a result of fishery gear interactions including hook and line and trap pot (A. Foley, Florida Fish and Wildlife Conservation Commission, pers. comm., 2013). This represents an increase from approximately 20 years prior, where 208 green turtles were reported stranded as a result of fishery gear interaction throughout the southeastern U.S. (Teas and Witzell, 1996).

5.2.6. Summary of Existing Conservation Efforts

The North Atlantic DPS is protected by various international treaties and agreements as well as national laws. As a result of these designations and agreements, many of the intentional impacts directed at sea turtles have been lessened; for instance, harvest of eggs and adults has been reduced at several nesting areas. The amount that these threats have been reduced as a result of these designations and agreements is not known.

The main threats to North Atlantic DPS green turtles include fishery bycatch mortality, particularly in gill net and trawl fisheries, nesting beach habitat loss and degradation via beachfront lighting and coastal armoring, and ingestion of marine debris during the epipelagic life stage. In addition, mortality from vessel strikes is increasing and likely a significant threat to this DPS. Mortality resulting from domestic and international commercial fishing ranks among the most significant threats to the North Atlantic DPS. Fishing gear types include gill nets, trawls, hook and line, seines, dredges, and various types of pots/traps. Among these, gill nets, hook and line, and trawl gear collectively result in thousands of North Atlantic green turtle deaths annually throughout their range (NMFS, 2002).

In the North Atlantic, nest protection efforts have been implemented on two major green turtle nesting beaches, and progress has been made in reducing mortality from human-related impacts on the other nesting beaches. In Costa Rica, the main green turtle nesting beach and by far the...
largest in the DPS, Tortuguero National Park, was established in 1976 to protect the habitat (Bjorndal et al., 1999). In Florida, a key effort was the acquisition of the Archie Carr National Wildlife Refuge in Florida in 1991, where nesting densities often exceed ranges from 36 nests per km (22 nests per mile) to 482 nests per km (77419 nests per mile) in the Refuge (D. Bagley, University of Central Florida, pers. comm., 2014; K. Kneifl, USFWS, pers. comm). Over 60 percent of the available beachfront acquisitions for the Refuge have been completed as the result of a multi-agency land acquisition effort. In addition, Hobe Sound National Wildlife Refuge, as well as coastal national seashores such as the Dry Tortugas National Park and Canaveral National Seashore, military installations such as Patrick Air Force Base and Canaveral Air Force Station, and State parks where green turtles regularly nest are also provided protection. However, despite these efforts, alteration of the coastline continues and, outside of publicly-owned lands, coastal development, and associated coastal armoring remains a serious threat.

Efforts are ongoing to reduce light pollution on U.S. nesting beaches. A significant number of local governments in the southeast U.S. have enacted lighting ordinances designed to reduce the effects of artificial lighting on sea turtles. However, enforcement of the lighting ordinances varies considerably. See Section 5.2.5.1 for a more complete discussion of this issue.

Considerable effort has been expended since the 1980s to document and reduce commercial fishing bycatch mortality. In the Atlantic and Gulf of Mexico, measures (such as gear modifications, changes to fishing practices, and time/area closures) are required to reduce sea turtle bycatch in pelagic longline, mid-Atlantic gill net, and southeast shrimp and flounder trawl fisheries. NMFS has implemented observer programs in many federally managed and some state-managed fisheries to collect turtle bycatch data and estimate mortality. NMFS, working with industry and other partners, has reduced bycatch in some fisheries by developing technological solutions to prevent capture or to allow most turtles to escape without harm (e.g., TEDs), by modifying gear (e.g., requirements to reduce mesh size in the leaders of pound nets) to prevent incidentally captured in shrimp trawl gear.

Since 1989, the U.S. has prohibited the importation of shrimp harvested in a manner that adversely affects sea turtles. The import ban does not apply to nations that have adopted sea turtle protection programs comparable to that of the U.S. (for example, require and enforce the use of TEDs) or to nations where incidental capture in shrimp fisheries does not present a threat to sea turtles (that is, nations that fish for shrimp in areas where sea turtles do not occur). The United States has required the use of TEDs throughout the year since the mid-1990s, with modifications required and implemented as necessary (52 FR 24244, June 29, 1987; 57 FR 57348; 57 FR 57348, December 4, 1992).

NMFS is currently working to implement a coastwide, comprehensive strategy to reduce bycatch of sea turtles in state and federal fisheries in the U.S. Atlantic and Gulf of Mexico. This approach was developed to address sea turtle bycatch issues on a per-gear basis, with a goal of developing and implementing coastwide solutions for reducing turtle bycatch inshore, nearshore, and offshore. The development and implementation of TEDs in the shrimp trawl fishery is arguably the most significant conservation accomplishment for North Atlantic green turtles in the
marine environment since their listing. In the southeast U.S. and Gulf of Mexico, TEDs have been mandatory in shrimp and flounder trawls for over a decade. However, TEDs are not required in all trawl fisheries, and significant green turtle mortality continues in some trawl fisheries. The estimated number of green turtles injured and/or killed by shrimp trawls each year in the Gulf and U.S. Southeast Atlantic combined is between 4,620 and 7,055 [NMFS, 2002].

In 2001, NMFS developed a comprehensive strategy to reduce bycatch of sea turtles in state and federal fisheries in the U.S. Atlantic and Gulf of Mexico. This approach was developed to address sea turtle bycatch issues on a per-gear basis, with a goal of developing and implementing coastwide solutions for reducing turtle bycatch inshore, nearshore, and offshore. Epperly and Teas (2002) indicated that 1 percent to 7 percent of stranded green turtles were too large to fit through the previous TED openings. In 2012, NMFS revised the TED requirement to increase maximum mesh size on escape flaps and the use of larger TEDs and Boone Wedge Cut escape openings (77 FR 29905, May 21, 2012). In addition, enforcement of TED regulations depends on available resources, and illegal or improperly installed TEDs continue to contribute to mortality.

Gill nets of various mesh sizes are used extensively to harvest fish in the Atlantic Ocean and Gulf of Mexico. All size classes of green turtles in coastal waters are prone to entanglement in gill nets, and, generally, the larger the mesh size the more likely that turtles will become entangled. State resource agencies and NMFS have been addressing this issue on several fronts. In the southeast U.S., gill nets are prohibited in the state waters of South Carolina, Georgia, Florida, and Texas and are restricted to fishing for pompano and mullet in saltwater areas of Louisiana. NMFS has addressed the issue for several federally managed fisheries, such as the large mesh gill net fishery (primarily for monkfish) along the Atlantic coast, where gill nets larger than 8-inch stretched mesh are now regulated in North Carolina and Virginia. The large mesh drift net fishery for sharks off the Atlantic coast of Florida and Georgia remains a concern as do gill net fisheries operating elsewhere in the range of the DPS, including Mexico and Cuba. In recent years, NMFS has dedicated significant funding and effort to address the bycatch issue. Although numerous efforts are underway to reduce green turtle bycatch in fisheries, and many positive actions have been implemented, this source of mortality is expected to continue across the range of the DPS because of the diversity and magnitude of the fisheries operating in the North Atlantic, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and lack of availability of comprehensive bycatch reduction technologies.

With regard to marine debris, the International Convention for the Prevention of Pollution from Ships, 1973, as modified by the Protocol of 1978 (MARPOL), is the main international convention that addresses prevention of pollution (including oil, chemicals, harmful substances in packaged form, sewage, and garbage) of the marine environment by ships from operational or accidental causes. However, challenges remain to implementation and enforcement of the MARPOL Convention, and on its own the Convention does not suffice to prevent all instances of marine pollution. The seriousness of the threat caused by vessel strikes to green turtles in the Atlantic and Gulf of Mexico cannot be overstated. This growing problem is particularly difficult to address. In some cases, NMFS, through Section 7 of the ESA, has worked with the U.S. Coast Guard in an attempt to reduce the probability of vessel strikes during permitted offshore race
events. However, most vessel strikes occur outside of these venues and the growing number of licensed vessels, especially inshore and nearshore, exacerbates the conflict. A number of regulatory instruments at international, regional, national, and local levels have been developed that provide legal protection for green turtles globally and within the North Atlantic Ocean.

With regard to addressing marine debris in the U.S., the Marine Debris Research, Prevention, and Reduction Act passed in 1987 (MDPRRA; 33 U.S.C. 1951 et seq.) The objective of the MDCC is to coordinate marine debris research, prevention, reduction, and removal activities among Federal agencies, in coordination with nongovernmental organizations, industries, universities, research institutions, States, Tribal Governments, and other countries.

The Wildlife Conservation Society in collaboration with local stakeholders through community and regional level meetings, have developed a management strategy for marine turtle conservation in Nicaragua. Each plan calls for a reduced level of take of green turtles and regulates markets and commercialization between regions and among communities. Regional and local authorities in Nicaragua are taking important steps towards making the Nicaragua green turtle fishery sustainable (Lagueux et al., 2012).

5.2.6.1. Regional and National Legislation and Protection

The 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 (SWOT, 2009).

Belize

In June 2002, the fisheries regulations were revised to prohibit fishing, possession, or trade in products of all 6 species of sea turtles found in the region. The regulations allow some fishing for "traditional" use (hawksbills cannot be taken under this usage) and require that shrimp trawlers use devices that let turtles escape from fishing equipment if they are caught accidentally (Searle, 2006).

Bermuda

The Protected Species Act of 2003 prohibits the take, import, export, sale, or purchase of a protected species. Green turtles were listed under the Amended Protected Species Order in 2012 (BR 7/2012).

Canary Islands

Cayman Islands

Green turtles were first protected in 1978 when regulations were put into place prohibiting possession of eggs and banning taking of nesting females from May through September (Cayman Islands Government, 1978). In 1996, the regulation was amended to prohibit 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. Licensing conditions stipulate size limits of no less than 40 and no more than 60 cm curved carapace length for legal take of green turtles (Cayman Islands Government, 2008).

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 green turtles.

Cuba

In 2008, the Ministry of Fishing Industries, Resolution 9, implementing the harvesting ban for all sea turtle species and products from its beaches and seas.

Dominican Republic

Decree No. 34-96 (1996). This decree established a five-year ban (1996-2001) on the capture, killing, collection, and commerce of green, hawksbill, loggerhead, and leatherback turtles, their eggs, and parts (Article 1) (Fleming, 2001[AML7]).

Guatemala

Ley General de Pesca y Acuicultura Decreto N° 80 (2002) Bräutigam and Eckert, 2006). These regulations were later confirmed in 2004. New regulations affecting the green turtles included controlling activities to curtail poaching and illegal trade of sea turtles and its eggs and the enforcement of TEDs in shrimp boats to reduce the number of accidental deaths (Arana, 2006).
**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. However, these regulations are ignored.

**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.

**Jamaica**

The Wildlife Protection Act was amended in 1991 (Fleming, 2001). Jamaica also passed the Endangered Species Act (Protection, Conservation and Regulation of Trade) to implement the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES, see section 4.3116.1.4.).

**Mauritania**

The capture, possession, sale and exportation of live wild animals are prohibited (1997).

**Mexico**

The most important law for sea turtle projection in Mexico was a 1990 presidential decree was proclaimed 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.

An additional law for sea turtle protection was a modification of the official Mexican Regulation NOM-002-PESC-1993 that was passed in 1997 to mandate the responsible management of shrimp fisheries throughout Mexico by implementing the use of turtle excluder devices. In 204 the Official Mexican Emergency Regulation NOM-EM-007-PESC was passed that provided technical specifications for the turtle excluder devices used by the shrimp trawling fleet in Mexico.

**Nicaragua**

The tradition of consuming turtle eggs is prohibited by law (Law No. 641 and Ministerial Resolution No. 043-2005). However, the harvesting and consumption of turtle eggs continue throughout the coastal areas.
**Panama**


**United States**

There are numerous laws and legislation in the United States that promotes the protection and conservation of sea turtles. The most relevant to sea turtle protection within U.S. Borders is the U.S. Endangered Species Act of 1973. The ESA has as its purpose 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. 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 conservation plan (often called HCP for 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. The ESA has been instrumental in curtailing the demise and assisting in the recovery of many species, and green turtles are included among them.

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 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 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 countries, with emphasis on protecting nesting populations and nesting habitat.

**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).
5.2.6.2. **International Instruments**

At least fifteen regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the North Atlantic Ocean. The international instruments listed below apply to sea turtles found in the North Atlantic Ocean, and their descriptions are given in Appendix 5.

- Convention on the Conservation of Migratory Species of Wild Animals
- Convention on International Trade in Endangered Species of Wild Fauna and Flora
- Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region
- Food and Agriculture Organization Technical Consultation on Sea Turtle-Fishery Interactions
- Inter-American Convention for the Protection and Conservation of Sea Turtles
- International Union for Conservation of Nature
- International Convention for the Prevention of Pollution from Ships (MARPOL)
- Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa.
- United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing
- United States Magnuson-Stevens Fishery Conservation and Management Act
- Western Hemisphere Convention

As a result of these designations and agreements, many of the intentional impacts directed at sea turtles have been lessened: harvest of eggs and adults has been reduced at several nesting areas through nesting beach conservation efforts and an increasing number of community-based initiatives are in place to reduce the take of turtles in foraging areas. In regard to incidental take, the implementation of TEDs has proved to be beneficial in some areas, primarily in the United States and South and Central America (National Research Council, 1990b). However, despite these advances, human impacts continue throughout the North Atlantic. The lack of effective monitoring in pelagic and near-shore fisheries operations still allows substantial direct and indirect mortality, and the uncontrolled development of coastal and marine habitats threatens to destroy the supporting ecosystems of long-lived green turtles.

5.3. **Assessment of Significant Portion of its Range (SPR)**

There are spatially explicit threats in this DPS (e.g., harvesting and bycatch issues outside of U.S. waters; coastal development, such as construction of sea walls, and high incidence of FP disease in Florida, US); however, no portion of the DPS range stands out as being at substantially greater risk of extinction than others. One potential exception is Cuba, but if the two rookeries therein were lost, it would not result in an increased risk of extinction to the DPS as a whole. Because the status of rookeries and the nature and degree of threats are relatively uniform across the range of the North Atlantic DPS, the SRT concluded that the need to consider a significant portion of the range does not apply to this DPS.
5.4. Assessment of Extinction Risk

For the SRT’s assessment of extinction risk for green turtles in the North Atlantic DPS, there were two separate sets of ranking exercises: One focusing on the importance that each SRT member placed on each of the six critical assessment elements for this region (Table 5.3), and a second which reflects the SRT members’ expert opinion about the probability that green turtles would fall into any one of various extinction probability ranges (Table 5.4). See Section 3.3. for details on the six elements and the voting process.

Table 5.3. Summary of ranks reflecting the importance placed by each SRT member on the critical assessment elements considered for the North Atlantic DPS. For Elements 1-4, higher ranks indicate higher risk factors.

<table>
<thead>
<tr>
<th>Critical Assessment Elements</th>
<th>Element 1 (Abundance 1 to 5)</th>
<th>Element 2 (Trends / Productivity 1 to 5)</th>
<th>Element 3 (Spatial Structure 1 to 5)</th>
<th>Element 4 (Diversity / Resilience 1 to 5)</th>
<th>Element 5 (Five-Factor Analyses 0 to 2)</th>
<th>Element 6 (Conservation Efforts 0 to 2)</th>
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<tr>
<td>MEAN RANK</td>
<td>1.18</td>
<td>1.18</td>
<td>1.45</td>
<td>1.36</td>
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<td>0.82</td>
</tr>
<tr>
<td>SEM</td>
<td>0.18</td>
<td>0.18</td>
<td>0.16</td>
<td>0.20</td>
<td>0.21</td>
<td>0.18</td>
</tr>
<tr>
<td>RANGE</td>
<td>1–3</td>
<td>1–3</td>
<td>1–2</td>
<td>1–3</td>
<td>(-2)–0</td>
<td>0–2</td>
</tr>
</tbody>
</table>

With respect to the important rankings for the six critical assessment elements, the average of the scores for the first four elements (Abundance, Productivity, Spatial Structure, and Diversity) was similar and relatively low, ranging from 1.18 to 1.45 in the risk threshold voting.

SRT members also generally thought that future threats not yet reflected in nester abundance or not yet experienced by the population weighed less in their risk assessment voting (average of -0.45) than did any conservation efforts that may emerge in the future (average of 0.82). SRT members had diverse opinions when considering the critical assessment elements. With respect to the diversity of opinions among the SRT members when considering the first four critical assessment elements, the largest range in rankings (i.e., voter opinion) was noted for Abundance, Trends / Productivity, and Diversity / Resilience sections (w/ ranks from 1 to 3). The diversity of opinions for threats and conservation efforts was reflected by the largest range of score possible for each.
Table 5.4. Summary of Green Turtle SRT member expert opinion on the probability that the North Atlantic DPS will reach a critical risk threshold within 100 years. Each SRT member assigned 100 points across the rank categories. The continuum in Row 1 has categories with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1-5%</th>
<th>6-10%</th>
<th>11-20%</th>
<th>21-50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN ASSIGNED POINTS</td>
<td>87.00</td>
<td>3.00</td>
<td>1.36</td>
<td>4.09</td>
<td>4.09</td>
<td>0.45</td>
</tr>
<tr>
<td>SEM</td>
<td>8.82</td>
<td>1.04</td>
<td>0.73</td>
<td>4.09</td>
<td>4.09</td>
<td>0.45</td>
</tr>
<tr>
<td>Min</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Max</td>
<td>100</td>
<td>10</td>
<td>7</td>
<td>45</td>
<td>45</td>
<td>5</td>
</tr>
</tbody>
</table>

Of the critical risk threshold categories describing the probability that the North Atlantic DPS will reach a critical risk threshold within 100 years (Table 5.4), SRT members voted overwhelmingly in the ‘<1%’ risk range (mean=87). The ‘1-5%’ and ‘6-10%’ categories had much lower average points (mean of 3.0 and 1.36, respectively). The ‘11-20%’ and the ‘21-50%’ categories each had a mean score of 4.09. The score decreased for the final category (‘>50%’), with a mean of 0.45. The range of scores in the individual risk categories was very high for the ‘<1%’ category, which ranged from 0 to 100.

In their vote justifications, SRT members cited the large geographic area of the DPS, with numerous nesting sites that have high abundance of nesters. Additional factors that were cited included the positives trends, particularly the PVA results. SRT members’ comments also included the increasing threat of coastal development and the continual threat of bycatch. There were concerns about climate change including loss of nesting beaches due to erosion and sea level rise.

It should be noted that, seven small nesting sites from the southeastern U.S. were considered in these votes by the SRT (see Table 5.1); however, because they represent only 0.4% of the estimated nester abundance and were within a DPS portion already containing major nesting sites, we don’t consider them to be of sufficient significance to trigger a new round of extinction risk voting.

5.5. Synthesis and Integration

During the consideration of the North Atlantic DPS’ status, an integrated approach was taken by the SRT to consider the many critical elements described earlier. The North Atlantic DPS is characterized by geographically widespread nesting with eight sites having high levels of abundance (i.e., <1,000 nesters). Nesting is reported in 16 countries and/or U.S. Territories at 74 sites. This region is data rich and has some of the longest running studies on nesting and foraging turtles anywhere in the world. All major nesting populations demonstrate long-term increases in abundance. A relatively low level of spatial structure is detected in this DPS due to
shared common haplotypes. The dispersed location of nesting sites provides a level of habitat use diversity and population resilience which reduces overall extinction risk.

Habitat loss due to coastal development and lighting issues are concerns in several areas, as well as the continued harvest of turtles in Nicaraguan waters and bycatch issues in Cuban waters.

The Five-Factor Analysis highlighted the continuing threats to green turtle habitat that affects all life stages of green turtles. Nesting beaches throughout some portions of the DPS are susceptible to coastal development and associated beachfront lighting, erosion, and sea level rise. Nesting and hatchlings are susceptible to predation which is prevalent throughout the beaches of the North Atlantic DPS and can be an increasing threat without nest protection and predatory control programs in place.

The extent of fishing practices and marine pollution is broad with high levels occurring in waters where high numbers of green turtles are known to forage and migrate. Along with degraded foraging and migratory areas, green turtles are also susceptible to direct harvesting. Current legal and illegal harvest of green turtles and eggs for human consumption continues in the eastern Atlantic and the Caribbean.

The prevalence of FP has reached epidemic proportions in some parts of the North Atlantic DPS. The extent to which this will affect the long-term outlook for green turtles in the North Atlantic DPS is unknown and remains a concern, although nesting trends across the DPS continue to increase despite the high incidence of the disease.

The long-term population trends and abundance of nesting females had the greatest influence on the SRT’s assessment of extinction risk and SRT members attributed the largest probability (87.0) to the lowest category of extinction risk (<1%). However, the characteristics of this DPS did lead one voter to conclude a 9% probability of having at least an 11% extinction risk. These results reflect the view that while the DPS shows strength in many of the critical assessment elements, there are still concerns about future risks, including habitat degradation (particularly coastal development), bycatch in fishing gear, continued turtle and egg harvesting, and climate change.
6. MEDITERRANEAN DPS (DPS #2)

6.1. DPS Range and Nesting Distribution

The Mediterranean Sea is a virtually enclosed basin occupying an area of approximately 2.5 million square kilometers. The Mediterranean Sea to the south separates Europe from Africa and the western boundary is the Atlantic Ocean. The Mediterranean DPS is bounded by the entire coastline of the Mediterranean Sea, but excluding the Black Sea. The western-most border of this DPS is marked by the Strait of Gibraltar.

Based on genetic data, green turtles are highly discrete from the wider Atlantic (Bowen et al., 1992; Encalada et al., 1996), and little if any interchange of individuals is present with the Atlantic. Nesting is concentrated in the eastern Mediterranean primarily in Cyprus, Turkey, and Syria, with lower levels of nesting in Syria, Lebanese Republic (Lebanon), Israel, and Egypt (Kasparek et al., 2001; Rees et al., 2008). Nesting is considered very rare to non-existent along the Mediterranean French and Spanish coasts, Malta, and the Hellenic Republic (Greece), and rare in Tunisia, and the Adriatic Sea (Italian Republic (Italy), Republic of Croatia (Croatia), and Republic of Albania (Albania; Kasparek et al. 2001). Data are not available or no records exist for Lebanon, Morocco, and Republic of Slovenia (Slovenia; Figure 6.1).

![Nesting distribution of green turtles in the Mediterranean DPS](image)

**Figure 6.1.** Nesting distribution of green turtles in the Mediterranean DPS (water body labeled '2'). Size of circles indicates nesting abundance category. Locations marked with 'x' indicate nesting sites lacking abundance information.

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**Comment [A2]:** This should be amended on the basis of the review by Casale & Margaritoulis (2010) and specifically the first chapter (Overview). Also note that mentioning Lebanon here contradicts what stated before.
Foraging areas and over-wintering habitats in the Mediterranean Sea have been proposed mainly though distributional information derived from the interaction of turtles with fishing gear (Margaritoulis et al., 2003). Juvenile green turtles have been recorded throughout the Mediterranean (Margaritoulis et al., 1992; Laurent et al., 1997; Meschini, 1997; Godley et al., 1998a, 1998b; Gianguzza et al., 2000; Oruç, 2001; Lazar et al., 2004), with apparent foraging grounds found in the eastern Mediterranean, in Lakonikos Bay, Greece (Margaritoulis et al., 1992; Margaritoulis and Tenekezis, 2003), off Fethiye Beach, Turkey (Türkozan and Durmuş, 2000), along the southeastern coast of Turkey near Syria (Yaşlıçay and Aureggi, 2006), and in Episkopi Bay, Cyprus (Stokes et al., 2011). Juvenile green turtles are frequently recorded in Libyan coastal waters (Ain al Ghazalah lagoon and along the coast between Sirte and Misratah), and to a lesser extent along the coast between Sirte and Misratah, Egypt probably hosts important foraging areas for green turtles as suggested by satellite tracking results from Cyprus and Syria, as well as high by-catch levels (Casale and Margaritoulis, 2010). Green turtles are occasionally found in the Adriatic Sea. Two stranded immature turtles (29.5 and 38.5 cm CCL) were found as far north as Lido Nazioni, Italy in the northwestern region (Vallini et al., 2011). Since 2003, at least 16 juvenile green turtles (27-67 cm CCL) were captured by fishermen using Stavnik (fish traps/weirs) in Albania (Haxhiu and Rumano, 2006 as cited in Casale and Margaritoulis, 2010). It has been suggested that this region may contain pelagic habitats for green turtles as well (Lazar et al., 2004).

6.2. Critical Assessment Elements

In the evaluation of extinction risk for green turtles in the Mediterranean DPS, the SRT considered six critical assessment elements, each of which are discussed below: (1) Nesting Abundance, (2) Population Trends (3) Spatial Structure, (4) Diversity / Resilience, (5) Five-Factor / Threat Analyses, and (6) Existing Conservation Efforts. See Section 3.3 for additional information on the selection of these six critical assessment elements.

6.2.1. Nesting Abundance

There are four nesting concentrations in the Mediterranean from which data are available, including those in Turkey, Cyprus, Israel, and Syria. Currently, approximately 452-2051 nests are laid in the Mediterranean each year - about 70 percent in Turkey, 15 percent in Cyprus, and 15 percent in Syria with trace nesting in Israel, Egypt, Greece, and Lebanon (Table 6.1; Casale and Margaritoulis, 2010). In terms of nester distribution among nesting sites in the Mediterranean, there were 32 sites, with the largest nesting site (Akyatan, Turkey) hosting 25 percent of the total annual nesting (Table 6.2).

The discovery of green turtle nesting in Syria in 2004 adds an average of 163 nests/yr (range 20-319) to green turtle nesting activity in the Mediterranean (Casale and Margaritoulis, 2010). That such a major nesting concentration could have gone unnoticed until recently (the Syrian coast was surveyed in 1991, but nesting activity was attributed to loggerheads (Carettacaretta)) bodes well for the ongoing speculation that the unsurveyed coast of Libya may also host substantial nesting.
Table 6.1. Summary of green turtle nesting sites in the Mediterranean DPS. Data are organized by country, nesting site, monitoring period, and estimated total nester abundance \[\text{(total counted females / year of monitoring) \times \text{remigration interval,}}\] and represent only those sites with sufficient data to estimate number of females. Many nesting sites in the Mediterranean DPS are data deficient and estimates could not be made for those beaches. For a list of references for these data, see Appendix 2.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Cyprus</td>
<td>North Karpaz, Region A</td>
<td>1998-2002</td>
<td>48</td>
</tr>
<tr>
<td>Cyprus</td>
<td>Alagadi, Region A</td>
<td>2009-2010</td>
<td>35</td>
</tr>
<tr>
<td>Cyprus</td>
<td>South Karpaz, Region A</td>
<td>2001-2002</td>
<td>12</td>
</tr>
<tr>
<td>Cyprus</td>
<td>West Coast, Region A</td>
<td>1993-2007</td>
<td>14</td>
</tr>
<tr>
<td>Cyprus</td>
<td>West Coast, Region B</td>
<td>2006-2008</td>
<td>20</td>
</tr>
<tr>
<td>Egypt</td>
<td>Egypt</td>
<td>1998</td>
<td>2</td>
</tr>
<tr>
<td>Greece</td>
<td>Greece</td>
<td>2007</td>
<td>1</td>
</tr>
<tr>
<td>Israel</td>
<td>Israel</td>
<td>1993-2008</td>
<td>3</td>
</tr>
<tr>
<td>Lebanon</td>
<td>El Aabbassiye</td>
<td>2003-2005</td>
<td>3</td>
</tr>
<tr>
<td>Lebanon</td>
<td>Tyre Coast Nature Reserve</td>
<td>2004-2005</td>
<td>1</td>
</tr>
<tr>
<td>Lebanon</td>
<td>El Mansouri</td>
<td>2002-2005</td>
<td>2</td>
</tr>
<tr>
<td>Syria</td>
<td>Latakia</td>
<td>2004-2009</td>
<td>6-91</td>
</tr>
<tr>
<td>Syria</td>
<td>Ras el Basit</td>
<td>2004-2009</td>
<td>1-4</td>
</tr>
<tr>
<td>Syria</td>
<td>Um Toyour</td>
<td>2004-2009</td>
<td>1-2</td>
</tr>
<tr>
<td>Syria</td>
<td>Wadi Kandil</td>
<td>2004-2009</td>
<td>1-4</td>
</tr>
<tr>
<td>Syria</td>
<td>Banias area</td>
<td>2004-2009</td>
<td>1-15</td>
</tr>
<tr>
<td>Turkey</td>
<td>Alata</td>
<td>2002-2006</td>
<td>7-66</td>
</tr>
<tr>
<td>Turkey</td>
<td>Kazanli</td>
<td>1988-2006</td>
<td>24-134</td>
</tr>
<tr>
<td>Turkey</td>
<td>Akyatan</td>
<td>1988-2006</td>
<td>36-245</td>
</tr>
<tr>
<td>Turkey</td>
<td>Sugozu</td>
<td>2004</td>
<td>71</td>
</tr>
<tr>
<td>Turkey</td>
<td>Samandag</td>
<td>1988-2010</td>
<td>5-207</td>
</tr>
<tr>
<td>Turkey</td>
<td>Patara</td>
<td>2001</td>
<td>1</td>
</tr>
<tr>
<td>Turkey</td>
<td>Fenike-Kumluca</td>
<td>1994</td>
<td>2</td>
</tr>
<tr>
<td>Turkey</td>
<td>Belek</td>
<td>1994-2006</td>
<td>1-3</td>
</tr>
</tbody>
</table>

1 The region of Cyprus under Turkish control
2 The region of Cyprus under Greek control
<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Turkey</td>
<td>Kizilot</td>
<td>1990</td>
<td>1</td>
</tr>
<tr>
<td>Turkey</td>
<td>Anamur</td>
<td>2006</td>
<td>1</td>
</tr>
<tr>
<td>Turkey</td>
<td>Goksu Delta</td>
<td>1991-2006</td>
<td>1-7</td>
</tr>
<tr>
<td>Turkey</td>
<td>Tuzla</td>
<td>2006</td>
<td>3</td>
</tr>
<tr>
<td>Turkey</td>
<td>Karatas</td>
<td>1989</td>
<td>1</td>
</tr>
<tr>
<td>Turkey</td>
<td>Agyatan</td>
<td>2006</td>
<td>1</td>
</tr>
<tr>
<td>Turkey</td>
<td>Yelkoma</td>
<td>1996</td>
<td>1</td>
</tr>
<tr>
<td>Turkey</td>
<td>Yumurtalik</td>
<td>2006</td>
<td>1</td>
</tr>
</tbody>
</table>

Table 6.2. Green turtle nester abundance distribution among nesting sites in the Mediterranean.

<table>
<thead>
<tr>
<th>NESTER ABUNDANCE</th>
<th># NESTING SITES DPS 2</th>
</tr>
</thead>
<tbody>
<tr>
<td>n/a</td>
<td>0</td>
</tr>
<tr>
<td>1-10</td>
<td>21</td>
</tr>
<tr>
<td>11-50</td>
<td>5</td>
</tr>
<tr>
<td>51-100</td>
<td>3</td>
</tr>
<tr>
<td>101-500</td>
<td>3</td>
</tr>
<tr>
<td>501-1000</td>
<td>0</td>
</tr>
<tr>
<td>1001-5000</td>
<td>0</td>
</tr>
<tr>
<td>5001-10000</td>
<td>0</td>
</tr>
<tr>
<td>10001-100000</td>
<td>0</td>
</tr>
<tr>
<td>&gt;100,000</td>
<td>0</td>
</tr>
</tbody>
</table>

TOTAL SITES | 32
TOTAL ABUNDANCE | 404-992
PERCENTAGE at LARGEST NESTING SITE | 25% (Akyatan, Turkey)

6.2.2. Population Trends

Although the Mediterranean DPS is depleted from historic levels (Kasparek et al., 2001), nesting data gathered since the early 1990s in Turkey, Cyprus, and Israel show no apparent decreasing trend. There are seven sites for which 10 years or more of recent data are available for annual nester abundance (the standards for representing trends in bar plot in this report) (Figure 6.2). Of these, only one site—West Coast, Cyprus—met our standards for conducting a PVA (Figure 6.3), and thus is not represented in the bar plots below. See Section 3.2 for more on data quantity and quality standards used for bar plots and PVAs. For a list of references on trend data, see Appendix 3.
Figure 6.2. Nesting data for green turtle sites in the Mediterranean DPS with greater than 10 yrs of recent monitoring data, although with some missed years. These include West Coast Cyprus (20 yrs), Akrotiri, Cyprus (17 yrs), North Carpaz, Cyprus (10 yrs), Israel (31 yrs), Akyatan, Turkey (17 yrs), Kazanli, Turkey (13 yrs), and Samandag, Turkey (11 yrs).

Of the six sites with at least 10 yrs of nesting abundance data (Figure 6.2), increasing trends were apparent for Israel, Samandag (Turkey), and to a lesser extent, Akrotiri (Cyprus) and Kazanli (Turkey). No apparent trend was present for North Carpaz (Cyprus). With respect to the Mediterranean green turtle population's status as stable/increasing but depleted relative to historic levels, this dynamic is particularly apparent along the coast of Palestine/Israel, where 300-350 nests were deposited each year in the 1950s (Sella, 1995) compared to a mean of 8 nests/yr from
1993 to 2008 (Casale and Margaritoulis, 2010); nesting in Israel has clearly increased over the last two decades.

PVA was one aspect of the Population Trend element and was conducted for nesting sites that had a minimum of 15 years of recent nesting data with an annual nesting level of more than 10 females (for more on data quantity and quality standards used, see Section 3.2). To assist in interpreting these PVAs, we indicate the probability of green turtle nesting populations declining to two different biological reference points, one using a trend-based and the other an abundance-based threshold. The trend-based reference point for evaluating population forecasts is half of the last observed abundance value, i.e., a 50% decline. The abundance-based reference point was a total adult female abundance of 300 females. Risk is calculated as the percentage of model runs that fall below these reference points after 100 years. This PVA modeling has important limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. For a full discussion of these PVAs and these reference points, see Section 3.2.

Figure 6.3. Stochastic Exponential Growth (SEG) Model Output for West Coast, Cyprus. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, gray-green dotted line is trend reference, and red dotted line is absolute abundance reference. Nesters were computed from nest counts using 4.5 nests/female (Demetropoulos and Hadjichristophorou 1989).

Comment [A5]: Ref needed here. Actually a lower value of 2.9-3.1 is estimated by Broderick et al 2002.
Comment [A6]: Response: Per nesting database: Alagadi: 1.9 to 3.1. (Broderick et al 2002)
Cyprus MU: 3

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For this population, the outputs of the PVA model based on 20 years (1989-2008) of nesting beach monitoring data indicate that there is a 33.3 percent probability that this population will fall below the trend reference point (50 percent decline) at the end of 100 years, and a 63.9 percent probability that this population falls below the absolute abundance reference (100 females/yr) at the end of 100 years. Of note for the PVA output of this DPS is the fact that the Absolute Abundance Biological Reference Point (BRP; red dotted line) is above the 50 percent decline BRP (grey-green dotted line); a feature unique to the PVA output for this DPS relative to all other DPSs around the globe. This is due to the fact that this population is substantially smaller than all other populations, with a total adult female abundance of 404-992 turtles. Thus, there is a relatively small overall decrease in females that is needed to reach the absolute abundance reference point of 300 females (i.e., 100 females/yr with remigration interval of 3 years).

6.2.3. Spatial Structure

When examining spatial structure for the Mediterranean DPS, the SRT examined three lines of evidence, including genetic data, flipper and satellite tagging, and demographic data.

Genetic sampling in the Mediterranean has been extensive and the coverage in this region is substantial considering the relative low population sizes of the nesting sites. Within the Mediterranean, rookeries are characterized by one dominant haplotype CM-A13 and show no population substructuring between several rookeries in Cyprus and Turkey (Bagda et al., 2012). MtDNA studies have identified 2 stocks: (1) Cyprus (Karpaz, North Cyprus and Lara Bay) and (2) Turkey (Akayatan, Alata, Kazanli, Samandag and Yumurtalik; Bagda et al., 2012). There are no studies of foraging grounds in the Mediterranean to show if turtles from other regions enter the Mediterranean to forage, but Mediterranean turtles have not been detected foraging outside the Mediterranean (e.g., Lahanas et al., 1998; Monzón-Argüello et al., 2010).

With respect to flipper tagging, despite years of tagging (Demetropoulos and Hadjichristophorou, 2010, 1995; Y. Kaska, Pamukkale University, pers. comm., 2013), few tag recoveries have been reported and this line of evidence did not feature significantly in SRT deliberations. However, satellite tracking was an important consideration. Of the 16 adult females tracked during post-nesting migrations (13 from Cyprus, 2 from Syria, 1 from Israel), most went to Libya (Misurata, western Gulf of Sirte; n=5) or the Gulf of Bomba (n=3) (UNEP-MAP, 2011). Post-nesting females migrate primarily along the coast from their nesting beach to foraging ground, increasing likelihood of interacting with fisheries (Broderick et al., 2002).

The demography of green turtles in the Mediterranean appears to be consistent among the various nesting assemblages (Broderick and Godley, 1996; Broderick et al., 2002). This consistency in parameters such as mean nesting size, inter-nesting interval, clutch size, hatching success, nesting season, and clutch frequency suggests a low level of population structuring in the Mediterranean. Nesters in the Mediterranean DPS are notably smaller than those found in other regions (Hirth, 1997). The mean CCL for nesters in Northern Cyprus and Turkey was found to be 88-96 cm. Hatching success varies widely from 9-100 percent with an average of
84.2 percent for areas with available information. Clutch size range varies widely from 23 to 199 eggs/nest with an approximate midpoint of 115.5 eggs/nest. Average clutch frequency is 3 [Broderick et al., 2002a]. Internesting interval varies from 10-16 days at Alagadi, Northern Cyprus (Broderick et al. 2002a) to nesting sites.

6.2.4. Diversity / Resilience

The components considered under this critical assessment element include the overall nesting spatial range, diversity of nesting season, diversity of nesting site structure and orientation (e.g., high vs. low beach face, insular vs. continental nesting sites), and the genetic diversity within the DPS. Components such as these are important considerations for assessing the potential impact of catastrophic events such as storms, sea level rise, and disease.

The overall spatial range of the population is limited. Green turtle nesting is found primarily in the eastern Mediterranean (Turkey, Syria, Cyprus, Lebanon, Israel, and Egypt) as well as Greece and Libya [Kasperek et al. 2001]. Occasionally green turtles are found in the Adriatic Sea (Italy, Croatia, and Albania), in Tunisia and very rarely in Malta and the western basin (Casale and Margaritoulis, 2010). The nesting season is similar throughout the DPS, with green turtles nesting from June to August [Broderick et al., 2002a]. The consistency of nesting season is consistent throughout this DPS [June to August] (Broderick et al., 2002a), thus limiting the temporal does not provide a buffering against climate change, in terms of impact to storms and other seasonal events.

The fact that turtles nest on both insular and continental sites suggests some degree of nesting diversity, but with the sites so close together the benefits of this diversity may be minimized. Mitochondrial DNA studies have identified two stocks (see Section 6.2.3); however, in general there is low population substructuring in the Mediterranean.

6.2.5. Analysis of Factors Listed Under ESA Section 4(a)(1)

Section 4(a)(1) of the ESA lists five factors (threats) that must be considered when determining the status of a species. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

The following information on these factors / threats pertains to green turtles found in the Mediterranean DPS.

6.2.5.1. Factor A: Destruction or Modification of Habitat or Range

Coastal development, beachfront lighting, and erosion resulting from sand extraction, negatively affect hatchlings and nesting turtles throughout this DPS. Fishing practices and marine pollution...
also affect the turtles throughout the DPS, with higher numbers of interactions occurring in waters where green turtles are known to forage and migrate. All life stages of green turtles are affected by habitat destruction in the neritic/oceanic zone.

Terrestrial Zone

In the Mediterranean, some nesting beaches have become severely degraded from a variety of activities. Destruction and modification of green turtle nesting habitat result from coastal development and construction, beachfront lighting, sand extraction, beach erosion, vehicular and pedestrian traffic, and beach pollution (Kasparek et al., 2001; Casale and Margaritoulis, 2010). These activities may directly impact the amount and suitability of nesting habitat available to nesting females and thus affect the nesting success of green turtles, as well as the survivability of eggs and hatchlings. Major green turtle nesting sites (i.e., nesting beaches with greater than 40 nests per year) within this DPS are located in the eastern Mediterranean at Alata, Kazanlı, Akyatan, Sugözü, and Samandağ beaches in Turkey; Latakia beach in Syria; and North Karpaz, Alagadi, Morphou Bay, and Lara/Toxeftra beaches in Cyprus (Kasparek et al., 2001; Casale and Margaritoulis, 2010); therefore, the following threats to the nesting habitat are mostly focused on these areas.

In Turkey, there has been an increasing demand for petroleum storage plants within the green turtle nesting region during the past decade resulting in degradation of nesting beaches. At Kazanlı beach, nesting habitat has been degraded by the construction of greenhouses in the dunes (Kasperek et al., 2001; Türkozan and Kaska, 2010). Coastal construction on Samandağ and Kazanlı beaches is also of concern, particularly from associated lighting and human activities on the beach (Türkozan and Kaska, 2010). In Syria, coastal development, particularly for tourism, is limited. However, even though the primary green turtle nesting beach at Latakia is relatively undeveloped, there is pressure from the Ministry of Tourism to develop this beach as well as other coastal areas (Rees et al., 2010). In Cyprus, the increased construction of beachfront hotels and other properties in some areas in recent years, as well as the associated increase in beachfront lighting and human activity on the beach, is decreasing the quality of nesting habitat (Demetropoulos and Hadjichristophorou, 2010; Fuller et al., 2010b). Furthermore, changes and improvements to infrastructure, such as roads, have led to the destruction of a north coast beach in Cyprus and some damage to other beaches (Fuller et al., 2010b).

As indicated above, coastal development is usually accompanied by artificial lighting. In the Mediterranean, disorientation of hatchlings due to beachfront lighting has been recorded and is of great concern in some areas. On Kazanlı beach in Turkey, light pollution from a soda-chrome factory, the town, and tourist facilities impacts the quality of the nesting habitat, as well as threatens nesting females and hatchlings (Kasperek et al., 2001). In spite of the limited coastal development at Latakia beach in Syria, lighting landward of the beach is a serious problem along the southern end, and numerous hatchlings have been documented as being misoriented and crawling away from the sea and into the dunes and field behind the beach (Rees et al., 2010).

Beach erosion and sand extraction also pose a problem to green turtle nesting habitat in the Mediterranean. In Turkey, sand mining and beach erosion have been identified as the most critical problems affecting green turtle nesting beaches (Türkozan and Kaska, 2010).
Researchers have reported beach erosion along a core nesting section of Kazanlı beach (Durmuş, 1998; Kasparek et al., 2001). A jetty, which was constructed in the 1980s and early 1990s, caused a significant amount of erosion to this core nesting beach and even though the jetty was completely removed in 2006, the beach has not yet recovered (Türkozan and Kaska, 2010; MEDASSET, 2013). On Samandağ beach, the illegal extraction of sand has been particularly destructive (Kasparek et al. 2001; Oruç et al., 2003 as cited in Türk ozan and Kaska 2010). Sand mining also occurs in some locations on Latakia beach in Syria (Rees et al., 2010). On Cyprus, the removal of large quantities of sand from Alagadi beach occurred in the past but is no longer a problem on this beach; however, small-scale sand removal does occur on other beaches (Fuller et al., 2010b).

Beach driving is a problem on some green turtle nesting beaches in the Mediterranean. In Syria, tractors and 4-wheel drive vehicles are regularly driven on some beaches (e.g., Latakia beach; Jony and Rees, 2008; Rees et al., 2010). In addition to direct impacts to green turtles from running into and injuring or killing nesting females and hatchlings or crushing nests, the operation of vehicles on the beach has been found to reduce the quality of nesting habitat by compacting the sand, which hinders nesting females from constructing nests and hatchlings trying to emerge from nests, and creating tire ruts that prevent or impede hatchlings from reaching the ocean following emergence from the nest (Jony and Rees, 2008). On the west coast of Cyprus, some problems with beach driving still exist on two beaches, but at a much smaller scale than in the past (Demetropoulos and Hadjichristophorou, 2010).

Human activity on the beach at night during the nesting season can reduce the quality of nesting habitat by deterring or disturbing nesting turtles and causing them to avoid otherwise suitable habitat. In addition, human foot traffic can make a beach less suitable for nesting and hatchling emergence by increasing sand compaction and creating obstacles to hatchlings attempting to reach the ocean (Hosier et al., 1981). Akyatan beach in Turkey, although located in a relatively remote location, has a substantial amount of tourists, visiting beaches during the summer months, may enter this protected beach from around the Tuzla area and thousands of tents are placed on the beach, particularly around the Tuzla area (Kasparek et al., 2001; Türk ozan & Kaska 2010). Kasanlı-Kazanlı beach also has a substantial amount of human usage for swimming, fishing, and other recreational activities both during the day and at night (Kasparek et al., 2001). In the center of Israel, intense human activity has been identified as the cause for low green turtle nesting success (Levy, 2010).

The eastern Mediterranean is exposed to high levels of pollution and marine debris, in particular the nesting beaches of Cyprus, Turkey, and Egypt (Camiñas, 2004). In Turkey, marine debris washing ashore is a substantial problem and has degraded nesting beaches, especially Akyatan and Samandağ beaches. In Syria, Jony and Rees (2008) reported that beaches contain a large amount of plastic litter that washes ashore or is blown in from dumps located in the beach dunes; this litter has been documented as accumulating in such large amounts that it can hinder nesting females from locating suitable nesting sites and cause emergent hatchlings to have difficulty crawling to the sea (Rees et al., 2010). Marine debris has also been a significant problem on some beaches in Cyprus, although organized beach clean-ups in recent years have greatly reduced the amount of litter on the beach (Demetropoulos and Hadjichristophorou, 2010; Fuller et al., 2010b).
Threats to habitat in the green turtle neritic and/or oceanic zones in the Mediterranean include fishing practices, marine pollution, and climate change. The degree of threat and overall impacts are described below.

Trawling occurs throughout the Mediterranean. However, green turtles mainly frequent the eastern Mediterranean, primarily off Turkey, Syria, Cyprus, Lebanon, Israel, Egypt, Greece, and Libya, but also occasionally occur off Italy, Croatia, Albania, and Tunisia and very rarely off Malta and the western basin of the Mediterranean (Casale et al., 2010). This fishing practice has the potential to destroy bottom habitat in these areas. Fishing methods affect neritic zones by not only impacting bottom habitat, including seagrasses that are present, and incidentally capturing turtles, but also by depleting fish populations and thus altering ecosystem dynamics. Although bottom trawling is the fishing practice that likely has the most dramatic impacts on seagrasses, other fishing practices such as dynamite fishing may be very destructive at a local level (Tudela, 2004). Although illegal, explosions at sea, likely due to dynamite fishing, have been reported off the coast of Syria (Saad, unpubl. data, as cited in Rees et al. 2010). Khalil et al. (2009) reported that dynamite fishing offshore of nesting beaches is common problem in Lebanon. Illegal dynamite fishing also occurs year round in Libya (Hamza, 2010). Further, the Mediterranean is a site of intense tourist activity, and corresponding boat anchoring also may impact green turtle foraging habitat in the neritic environment. Climate change also may result in future trophic changes, including changes in the distribution, amount, and types of seagrasses and macroalgal species (Lapointe, 1999; Harley et al., 2006; Björk et al., 2008), thus altering green turtle foraging habitat (Hawkes et al., 2009).

Marine pollution, including direct contamination and structural habitat degradation, can affect green turtle neritic and oceanic habitat. As the Mediterranean is an enclosed sea, organic and inorganic wastes, toxic effluents, and other pollutants rapidly affect the ecosystem (Camínhas, 2004). The Mediterranean 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). 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 (UNEP, 2007).

Indirect effects can result from both point and non-point source pollution associated with coastal development (e.g., discharge of chemical substances from a soda-chromium factory close to the Kazanlı nesting beach in Turkey). The impacts of climate change may also result in trophic level alterations, and therefore may affect forage quantity, quality, and/or distribution.

6.2.5.2. **Factor B: Overutilization**

The harvesting of eggs and turtles was likely a factor that contributed to the historical declines of the population, and still occurs within a portion of this DPS.
Egg Harvest

Egg collection (for individual consumption) still occurs in Egypt (Clarke et al., 2000; Nada and Casale, 2008). In Cyprus, no recent incidences of exploitation of eggs have been reported (Demetropoulos and Hadjichristophorou, 2010; Fuller et al., 2010b). In Syria, we found no evidence of exploitation of green turtle eggs, nor any evidence of existing commercial enterprise for the meat of nesting turtles.

Turtle Harvest

Eastern Mediterranean sea turtle populations were subject to severe exploitation until the mid-1960s (Margaritoulis et al., 2003). Deliberate hunting of green turtles for their meat, blood, shells, and eggs is reduced from previous exploitation levels, but still exists. In the mid-1990s in Egypt, turtles were still being sold in fish markets despite prohibitive laws. Of 71 turtles observed at fish markets in 1995 and 1996, 32 percent were green turtles (Laurent et al., 1996). Nada (2001) reported 135 turtles (of which 15 percent were green turtles) slaughtered at the fish market of Alexandria in 6 months (December 1998–May 1999). Based on observed sea turtle slaughters in 1995 and 1996, Laurent et al. (1996) estimated that several thousand sea turtles were probably killed each year in Egypt. More recently, a study found that the open selling of sea turtles in Egypt generally has been curtailed due to enforcement efforts, but a high level of intentional killing for the black market or for direct personal consumption still exists (Nada and Casale, 2008). Given the high numbers of turtles caught in this area, several hundred turtles are currently estimated to be slaughtered each year in Egypt (Nada and Casale, 2008). This estimate likely includes both juvenile and adult loggerhead and green turtles, as Egyptian fish markets have been documented selling different sized sea turtles. While the mean green turtle size in the 1995–1996 study was 66.8 cm CCL (range 28–95.5 cm CCL; n=21), 19 percent of observed green turtle samples were 70 cm CCL or larger (Laurent et al., 1996).

Based on stranding records, Demetropoulos and Hadjichristophorou (2010) estimate one or two green turtles may be incidentally taken and killed by fishermen in western Cyprus annually. Similar taking of green turtles by fishermen in Greece has also been reported. Based on a sample of 226 turtles (of which 3.5 percent were green turtles), over 34 percent had turtles were recorded with head traumas attributed to intentional hits after incidental capture in fishing gear (Panagopoulos et al., 2003). A subsequent report with a larger sample size of 469 turtles raised the percentage of recorded a higher percent of turtles intentionally injured turtles to 41.6 percent (Panagopoulou et al., 2008).

In Syria and Egypt, as reported for other countries, green turtles incidentally captured by fisherman are sometimes eaten (Nada and Casale, 2008; Rees et al., 2010). Small quantities of stuffed turtles and juvenile turtle carapaces, presumably of Syrian origin, have been observed for sale in Latakia and Damascus (Rees et al., 2010).

6.2.5.3. Factor C: Disease or Predation

Disease was not found to be a factor that contributed to the historical decline of this DPS, while nest and hatchling predation likely was a factor that contributed to the historical decline of this
DPS. The best available data suggest that current nest and hatchling predation on several Mediterranean nesting beaches is a continued threat to this DPS.

The potential exists for diseases and endoparasites to impact green turtles in the Mediterranean. However, there have been no records of FP or other diseases in green turtles in this DPS. Nada and Casale(2008) conducted a rapid assessment of the presence and status of FP in Egypt through interviews with fishermen; however, none of the fishermen interviewed had ever encountered green turtles with the disease. Therefore, while there is the potential for disease in the Mediterranean, information on the presence and prevalence of such disease is lacking.

In the Mediterranean Sea, green turtle eggs and hatchlings are subject to depredation by wild canids (i.e., foxes (*Vulpes vulpes*), golden jackals (*Canis aureus*)), feral and domestic dogs (*Canis lupus familiaris*), and ghost crabs (*Ocypode cursor*) (van Piggelen and Strijbosch, 1993; Brown and MacDonald, 1995; Aureggi *et al.*, 1999, 2005; Simms *et al.*, 2002; Akcinar *et al.*, 2006; Jony and Rees, 2008; Khalil *et al.*, 2009; Aureggi and Khalil, 2010; Demetropoulos and Hadjichristophorou, 2010; Fuller *et al.*, 2010; Rees *et al.*, 2010).

Nest predation by canids is very common on Turkey’s nesting beaches. On Akyatan beach, green turtle nest predation by canids was reported at 63.8 percent in 1992 (Brown and MacDonald, 1995) and 23.8 percent in 1995 (Aureggi *et al.*, 1999). During the 2006–2009 nesting seasons, 170 to 562 green turtle nests were laid annually on Akyatan beach, and egg predation by jackals ranged from 14 to 25 percent annually (Türkozan *et al.*, 2011). Peters and Verhoeven (1992 as cited in Türkozan and Kaska 2010) reported that jackals have an even greater impact on hatchling survival than on nests. On Göksu Delta beach, jackals have been reported to kill nesting females (Akcinar *et al.*, 2006).

Egg and hatchling predation by dogs and other canids is also a major concern at Latakia beach in Syria (Rees *et al.*, 2010). Along the northern Cyprus coastline, predation is the most critical threat to sea turtle reproductive success, with nest predation by feral dogs and foxes reaching as high as 38 percent (includes both green and loggerhead turtle nests; mean 17.7 percent, range 8–38 percent) in a single year (Fuller *et al.*, 2010b). Along the western Cyprus coastline, fox predation historically reached 80 percent (includes both green and loggerhead turtle nests) on some beaches (Demetropoulos and Hadjichristophorou, 2010). However, nest predation on Cyprus has been greatly reduced (less than five percent on the western coastline) with the implementation of nest screening. In Lebanon, where green turtles only nest in small numbers, nest predation by foxes, jackals, dogs, and feral hogs has been observed on the southern beaches (Aureggi *et al.*, 2005).

Diperan larvae have been reported as infesting green turtle nests on Cyprus beaches (Broderick and Hancock, 1997; McGowan *et al.*, 2001). In 1996 and 1997, at least 3.3 percent and 20.7 percent, respectively, of green turtles nests were infested (McGowan *et al.*, 2001). However, most infestations likely involved moribund eggs, thus not posing a major threat to nests.

Ghost crab (*Ocypode sp.*) predation has been documented on Cyprus beaches, although this does not appear to be a significant threat (Demetropoulos and Hadjichristophorou, 2010; Fuller *et al.*, 2010b). In Egypt, however, Simms *et al.* (2002) observed a high level of predation by ghost
crabs, with levels ranging between 45 and 99 percent (includes both green and loggerhead turtles) on different beaches. Ghost crabs are abundant at Latakia beach in Syria and are likely sustained by the substantial amount of trash on the beach; they are responsible for a significant amount of hatchling predation on this beach (Jony and Rees, 2008). At El Mansouri beach in Lebanon, ghost crab predation on hatchlings prior to or during nest emergence or during their crawl to the ocean has been documented (Khalil et al., 2009).

6.2.5.4. Factor D: Inadequacy of Existing Regulatory Mechanisms

Regulatory mechanisms are in place that should address direct and incidental take of green turtles in the Mediterranean DPS; however, in some countries these regulatory mechanisms may not be implemented effectively or the regulations do not provide sufficient protection for all life stages of green turtles. Inadequacy of regulatory mechanisms may be a contributing factor to why impacts to the nesting beach habitat (Factor A), overutilization (Factor B), predation (Factor C), and fishery bycatch (Factor E) continue throughout the DPS to varying degrees.

There are at least 13 international treaties and/or regulatory mechanisms that pertain to the Mediterranean, and nearly all countries lining the Mediterranean have some level of national legislation directed at sea turtle protection (see Appendices 3 and 4). Hykle (2002) and Tiwari (2002) reviewed the value of some international instruments, which vary in their effectiveness. Often, international treaties 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, 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 a 2002 special issue of the Journal of International Wildlife Law and Policy: International Instruments and Marine Turtle Conservation (Hykle, 2002).

Fishery bycatch occurs throughout range of the Mediterranean DPS (Factor E). Anthropogenic threats to nesting beaches (Factor A) and eggs/hatchlings (Factors A, B, C, and E), are also substantial. Although conservation efforts to protect some nesting beaches are underway, more widespread and consistent protection is needed. While national and international governmental and non-governmental entities in the Mediterranean region are currently working toward reducing green turtle bycatch, it is unlikely that this source of mortality can be sufficiently reduced across the range of the DPS in the near future because of the lack of bycatch reduction in commercial and artisanal fisheries operating within the range of this DPS, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and lack of availability of comprehensive bycatch reduction technologies.

6.2.5.5. Factor E: Other Natural or Manmade Factors

Fishery bycatch that occurs throughout the range of the Mediterranean DPS, particularly bycatch mortality of green turtles from pelagic longline, set net, and trawl fisheries, is a continued threat to this DPS. Additional threats from boat strikes (which are becoming more common), power
generation, marine pollution, changes likely to result from climate change, and natural disasters will negatively affect this DPS.

6.2.5.5.1. Incidental Bycatch in Fishing Gear

Incidental capture of sea turtles in artisanal and commercial fisheries is a significant threat to the survivability of green turtles in the Mediterranean. Fishing practices alone have been estimated to result in over 150,000 sea turtle captures per year, with approximately 50,000 mortalities (Casale, 2008–2011; Lucchetti and Sala, 2009) and sea turtle bycatch in multiple gears in the Mediterranean is considered among the most urgent conservation priorities globally (Wallace et al., 2010b). Green turtles may be caught in pelagic longlines, set nets (gill nets and trammel nets), bottom and mid-water trawls, seines, and hook and line gear. In a 2004 FAO Fisheries Report, Camiñas (2004) stated that the main fisheries affecting sea turtles in the Mediterranean Sea (at that time) were Spanish and Italian longline, North Adriatic Italian, Tunisian, and Turkish trawl, and Moroccan and Italian drift net. Available information on sea turtle bycatch by gear type is discussed below. There is growing evidence that artisanal/small vessel fisheries (set gill net, bottom longline, and part of the pelagic longline fishery) may be responsible for a comparable or higher number of captures with higher mortality rates than the commercial/large vessel fisheries (Casale, 2008).

Longline Fisheries

In the Mediterranean, surface longline fisheries are a source of green turtle bycatch (Camiñas, 2004). Incidental captures have been reported from Cyprus (Godley et al., 1998b), Turkey (Godley et al., 1998b), Italy (Laurent et al., 2001), and Egypt (Nada, 2001; Camiñas, 2004). A survey of 54 small boat (4–10 meter length) artisanal fishermen in northern Cyprus and Turkey resulted in an estimated minimum bycatch of over 2,000 turtles per year, with an estimated 10 percent mortality rate (Godley et al., 1998b). These small boats fished with a combination of longlines and trammel/gill nets. It is likely that a large proportion of the turtle bycatch estimated in this study were juvenile green turtles (Godley et al., 1998b).

In Egypt, based on fleet data and catch rates reported by fishermen during the 2000s, the total number of sea turtles bycaught in longlines was estimated to be over 2,200 per year (Nada and Casale, 2008). Fishermen also reported that some of the caught turtles are dead and the incidence of mortality is particularly high in longlines and gill nets. Although the turtle numbers presented by Nada and Casale (2008) could not be broken down by species, Nada (2001) reported that of 135 turtles observed over a 6-month period in 1998–1999 for sale at the Alexandria Fish Market, 15 percent were green turtles.

Out of 200 sea turtles captured in pelagic longlines in Italian waters in the Ionian Sea during 1999 and 2000, only two were green turtles (Deflorio et al., 2005). Of 85 turtles captured by longlines in the south Adriatic, Ionian, Central Mediterranean, and Tyrhenian Seas, none were green turtles (Guglielmi et al., 2000, as cited in Casale, 2010). Based on this information and the fact that green turtles were not observed in turtle captures by bottom trawlers in either the north Adriatic Sea (Casale et al., 2004) or the Gulf of Gabes (Casale et al., 2007), Casale (2010) concluded that the Italian pelagic longline fishing fleet has a limited impact on green turtle due to the low occurrence of the species in Italian waters.
**Set Net (Gill net) Fisheries**

As in other areas, sea turtles have the potential to interact with set nets (gill nets or trammel nets) in the Mediterranean. Mediterranean set nets refer to gill nets (a single layer of net) and trammel nets, which consist of three layers of net with different mesh size. Casale(2008) estimated that the countries with the highest number of sea turtle captures (in the thousands per year) are Tunisia, Libya, Greece, Turkey, Cyprus, and Croatia. Italy, Morocco, Egypt, and France likely have high capture rates as well. Available information suggests the annual number of sea turtle captures by Mediterranean set nets may be greater than 30,000 (Casale, 2008); however, the number was not broken down by species.

Due to the nature of the gear and fishing practices (e.g., relatively long soak times), incidental capture in gill nets is among the highest source of direct sea turtle mortality. Considering data throughout the entire Mediterranean, as well as a conservative approach, Casale(2008) considered mortality by set nets to be 60 percent, with a resulting estimate of 16,000 turtles killed per year. Most of these animals are likely juveniles; Casale(2008) evaluated available set net catch data throughout the Mediterranean and found an average size of 45.4 cm CCL (n=74). However, a breakdown of these estimates by turtle species is not available.

In northern Cyprus, there is considerable turtle bycatch by artisanal fishermen using a combination of longlines and gill nets/trammel nets from approximately 180 fishing vessels. The estimated median number of green and loggerhead turtles captured by fisherman is four turtles per boat per year, with a 10 percent mortality rate (Godley et al., 1998b; Fuller et al., 2010). However, the number of turtles caught was not broken down by species because most fishermen were unable to distinguish between the two species. In western Cyprus, bottom set nets (trammel nets) pose the greatest fisheries bycatch problem for sea turtles. In Chrysochou Bay, an important foraging area for juvenile and adult green turtles in western Cyprus, approximately 20–30 dead juvenile green turtles were found stranded each year in 2006–2007. The strandings were documented most frequently during the summer months when fishing activities in Chrysochou Bay are more intensive (Demetropoulos and Hadjichristophorou, 2010).

In Turkey, there is also considerable turtle bycatch by artisanal fishermen using a combination of longlines and gill nets and trammel nets from approximately 530 fishing vessels. The estimated median number of green and loggerhead turtles captured by fisherman is 2.5 turtles per boat per year, with a 10 percent mortality rate (Godley et al., 1998b). Although the number of turtles caught was not broken down by species because most fishermen were unable to distinguish between the two species, analysis of 2002–2009 stranding data for the eastern Mediterranean coast of Turkey revealed that 46.7 percent of stranded dead turtles were green turtles (Türkozan et al., 2013).

Although the extent of fisheries interactions with sea turtles has not been fully assessed in Greece, in Lakonikos Bay, an area with 40 percent green turtles and 60 percent loggerheads, approximately 30 percent of 24 turtles caught in set nets were green turtles (Margaritoulis and Teneketzis, 2003). In Egypt, based on fleet data and catch rates reported by fishermen during the 2000s, the total number of sea turtles bycaught in set nets (gill nets) was estimated to be over 800
Fishermen also reported that some of the caught turtles are dead and the incidence of mortality is particularly high in longlines and gill nets. Although the turtle numbers presented by Nada and Casale (2008) could not be broken down by species, Nada (2001) reported that of 135 turtles observed over a 6-month period in 1998–1999 for sale at the Alexandria Fish Market, 15 percent were green turtles. In Croatia, only one out of 100 turtles handled during a 15-year period by the Adriatic Marine Turtle Research and Conservation Program was a green turtle; this turtle was captured dead in a gill net (Lazar et al., 2004; Lazar, 2010). Due to the low occurrence of green turtles in Croatian waters, fishery bycatch is likely to be relatively low for this species.

**Trawl Fisheries**

Green turtles have been reported as incidentally captured in bottom trawls in Egypt (Nada and Casale, 2011), Greece (Margaritoulis et al., 2003), Tunisia (Laurent et al., 1990), and Turkey (Laurent et al., 1996; Oruç, 2001), as well as Syria, Israel, and Libya (Casale et al., 2010) but are likely also captured by bottom trawlers in other neritic foraging areas in the eastern Mediterranean (Casale et al., 2010). Laurent et al. (1996) estimated that approximately 10,000 to 15,000 sea turtles were being captured annually by bottom trawling in the eastern Mediterranean. Although most of the turtles taken were loggerheads, they estimated that the number of green turtles taken was 1,000 to 3,000 annually in Turkey and Egypt alone. More recently, Casale (2011) compiled available trawl bycatch data throughout the Mediterranean and reported that Italy and Tunisia have the highest level of sea turtle bycatch, potentially over 20,000 captures per year combined, and Croatia, Greece, Turkey Libya, Greece, and Egypt each have an estimated 1,900 or more sea turtle captures per year. Further, Albania, Algeria, Cyprus, Morocco, Slovenia, Spain, and Syria may each capture a few hundred sea turtles per year (Casale, 2011). Available data suggest the annual number of sea turtle captures by all Mediterranean trawlers may be greater than 39,000 (Casale, 2011). Although most of the turtles reported by Casale (2011) as taken by bottom trawlers were undoubtedly loggerheads, a few thousand were likely green turtles based on earlier reports (Laurent et al., 1996, 1990; Oruç, 2001; Margaritoulis et al., 2003; Nada and Casale, 2008).

Based on available information from multiple areas of the Mediterranean, and assuming that comatose animals die if released in that condition, the overall average mortality rate for bottom trawlers was estimated to be 20 percent (Casale, 2011). Thus, over 8,000 turtles are estimated to be killed per year by bottom trawlers in all of the Mediterranean (Casale, 2011). However, it is important to note that trawl-induced mortality varies depending on a number of factors and may vary greatly between and within countries. A key factor affecting the mortality rate in trawls is the duration of the haul, with longer haul durations resulting in higher mortality rates (Henwood and Stuntz, 1987; Sasso and Epperly, 2006).

Mid-water trawling may have less total impact on sea turtles found in the Mediterranean than some other gear types, but interactions still occur. Off Turkey, 249 green turtles were captured in mid-water trawls during the 1996–1997 trawling season, while 30 green turtles were incidentally taken in bottom trawls (Oruç, 2001). In this same study, of a total 320 turtles captured in mid-water trawls (loggerheads and greens combined), nearly 95 percent were captured alive and apparently healthy. While the total catch numbers throughout the
Mediterranean have not been estimated, mid-water trawl fisheries do present a threat to green turtles.

Other Gear Types

Beach seine and weir fisheries, as well as other gear types, that operate in Mediterranean waters may also affect green turtles, although incidental captures in some of these gear types are largely unknown (Camíñas, 2004). Artisanal fisheries using a variety of gear types also have the potential for sea turtle takes, but the effects of most artisanal gear types on sea turtles have not been estimated.

Beach seines have been reported as capturing green turtles in Syria and Greece. Off Latakia beach in Syria, juvenile green turtles were captured in beach seines with up to nine turtles captured in a single setting of a single net (Rees et al., 2010). Observers noted that all turtles were released unharmed by fishermen back to the sea. Rees et al. (2010) concluded that if turtles captured in beach seines are not opportunistically exploited by fishermen, then the impacts of beach seines would not be expected to significantly impact Syria’s green turtle population. At Lakonikos Bay in Greece, an area with 40 percent green turtles and 60 percent loggerheads, approximately 84 percent (64) of 139 of turtles caught in beach seines were green turtles (Margaritoulis and Teneketzis, 2003). Although beach seine gear is gradually being withdrawn from Greek fisheries by non-renewal of licenses, Margaritoulis and Panagopoulou (2010) indicated that this gear is still taking a heavy toll on sea turtles in some areas.

In Albania, a type of fishing weir known as a Stavnik was documented as incidentally catching 16 green turtles since 2003 (Haxhiu, 2010). However, the Stavnik is reported to be a good fishing gear for sea turtles because the configuration of the gear allows non-target species to be returned to the sea unharmed (Haxhiu and Rumano, 2006a, 2006b).

6.2.5.5.2. Vessel Strikes

Propeller and collision injuries from boats and ships are becoming more common in sea turtles in the Mediterranean, although it is unclear as to whether the events are increasing or just the reporting of the injuries. Speedboat and jet-ski impacts are of particular concern in areas of intense tourist activity, such as Greece and Turkey. Boats operating near sea turtle nesting beaches during the nesting season are likely to either cause females to abandon nesting attempts or cause their injury or death (Camíñas, 2004). Males may also be impacted in high-use boating areas where sea turtle mating occurs (Demetropoulos, 2000).

An analysis of sea turtle strandings along the Greece coastline from 1997–1999 revealed that boat strikes were a seasonal phenomenon occurring primarily during the summer when tourist activity was highest (Kopsida et al., 2002). During this study, 9 percent of 524 turtles stranded along the coasts of Greece had injuries likely caused by boat strikes (Margaritoulis, 2007). Although the numbers of stranded green turtles affected by boat strikes were not presented in this study, all or the majority were juveniles (Kopsida et al., 2002). In northern Cyprus, increased tourism has also resulted in increased speedboat and jet ski usage in marine habitats, and in recent years, boat strikes of sea turtles have been reported, particularly in the Girne (Kyrenia)
area (Fuller et al., 2010b). In Syria, concerns have been raised about power boat and jet ski impacts, both from direct strikes and general disturbance, on green turtles at a foraging area near BerjEslam and Ibn Hani (Rees et al., 2010).

6.2.5.5.3. Power Generation Activities

In 2012, two licensed wind energy plants existed and applications for 28 additional plants were under consideration in Samandağ, Turkey, one of the most important green turtle nesting beaches in the Mediterranean (Yalçın-Özdilek and Yalçın, 2012). Although no information is available for green turtles, the entrainment and entrapment of a loggerhead sea turtle documented in the waterway of a state power plant station at Keratsili, Greece (Margaritoulis and Panagopoulou, 2010) demonstrates the potential for capture of green turtles by such systems in the Mediterranean. Although the loggerhead at this facility was eventually rescued, the potential for mortality exists.

6.2.5.5.4. Pollution

Direct or indirect disposal of anthropogenic debris introduces potentially lethal materials into green turtle foraging habitats. Unattended or discarded nets, floating plastics and bags, and tar balls are of particular concern in the Mediterranean (Camíñas, 2004; Margaritoulis, 2007). Monofilament netting appears to be the most dangerous waste produced by the fishing industry (Camíñas, 2004).

Contaminants in the marine environment may impact green turtles, although not to the extent they are likely to impact loggerheads due to differences in their dietary preferences (Godley et al., 1999). Even so, concentrations of contaminants from sea turtles in Mediterranean waters were found to be comparable to elsewhere in the Atlantic and areas in the North Pacific (Godley et al., 1999; Mckenzie et al., 1999), but the levels were much lower than the concentrations shown to cause deleterious effects in freshwater turtles (Mckenzie et al., 1999). In addition, contaminant burdens in green turtles were found to be highest in juveniles and are believed to decrease as they grow due to a decrease in contaminant intake (Mckenzie et al., 1999) as turtles shift from an omnivorous to an herbivorous diet. However, the discharge of chemical substances, including highly toxic chromium compounds, from a soda-chromium factory close to the Kazanlı nesting beach in Turkey is cause for concern (Kasperek et al., 2001; Venizelos and Kasperek, 2006).

6.2.5.5.5. Climate Change

Similar to other areas of the world, climate change and sea level rise have the potential to affect green turtles in the Mediterranean. As described in section 6.1.5., over the long term, Mediterranean turtle populations could be threatened by the alteration of thermal sand characteristics (from global warming), resulting in the reduction or cessation of male hatchling production (Kasperek et al., 2001; Camíñas, 2004; 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 hatchlings. In northern
Cyprus, green turtle hatchling sex ratios are already thought to be highly female biased (approximately 95 percent female; Wright et al., 2012). This, in tandem with predicted future rises in temperatures is cause for concern (Fuller et al., 2010b). As temperatures increase, there is also concern that incubation temperatures will reach levels that exceed the thermal tolerance for embryonic development, thus increasing embryo and hatching mortality (Fuller et al., 2010b). Thus, climate change impacts could have profound long-term impacts on green nesting in the Mediterranean, but it is not possible to project the impacts at this point in time. Further, a significant rise in sea level would restrict green turtle nesting habitat in the eastern Mediterranean.

6.2.5.6. Natural Disasters

Natural environmental events also may affect green turtles in the Mediterranean. Cyclonic storms that closely resemble tropical cyclones in satellite images occasionally form over the Mediterranean Sea (Emanuel, 2005). While hurricanes typically do not occur in the Mediterranean, researchers have suggested that climate change could trigger hurricane development in this area in the future (Gaertner et al., 2007). Any significant storm event that may develop could disrupt green turtle nesting activity and hatching production, but the results are generally localized and rarely result in whole-scale losses over multiple nesting seasons. However, when combined with the effects of sea level rise, there may be increased cumulative impacts from future storms.

6.2.6. Summary of Existing Conservation Efforts

Most Mediterranean countries have developed national legislation to protect sea turtles and nesting habitats (Casale and Margaritoulis, 2010). National protective legislation generally prohibits intentional killing, harassment, possession, trade, or attempts at these (e.g., Margaritoulis et al., 2003). In addition, some countries have site-specific legislation or conservation designation for turtle habitat protection.

In Turkey, three important green turtle nesting beaches (Alata, Kazanlı, and Akyatan) were all designated as protected areas by the Turkish Ministry of Culture, while two other beaches (Belek and Gösku Delta) also have some level of protected status (Kasperek et al., 2001; Fuller et al., 2010). These five protected beaches represent approximately 60 percent of nesting in Turkey (see Canbolat et al., 2009 and Fuller et al., 2010). In western Cyprus, Lara-Toxeftra beaches have been afforded protection through the Fisheries Law and Regulations since 1989 (Margaritoulis, 2007). In northern Cyprus, four beaches (Alagadi Beach, Karpaz Peninsular, South Karpaz, and Akdeniz) have been designated as Special Protected Areas (Fuller et al., 2010b); these four areas include the third and fifth most important green turtle nesting beaches in the Mediterranean (Kasperek et al., 2001). In Syria, establishment of a protected area at Latakia beach, the most important green turtle nesting beach in the country, is being sought but is facing strong opposition from the tourism sector (Rees et al., 2010). In summary, Mediterranean green turtle nesting primarily occurs in Turkey, Cyprus, and Syria, and a notable proportion of nesting in those areas is protected through various mechanisms. It is important to recognize the success of these protected areas, but as the protection has been in place for some time and the threats to the
species remain (particularly from increasing tourism activities), it is unlikely that the protective measures discussed here are sufficient for the conservation of the species in the Mediterranean.

Protection of marine habitats is in the early stages in the Mediterranean, as in other areas of the world. Off the Lara-Toxeftra nesting beaches in western Cyprus, a marine protection zone extends to the 20 m isobath as delineated by the Fisheries Regulation (Margaritoulis, 2007; Demetropoulos and Hadjichristophorou, 2010). As mentioned above, establishment of a protected area at Latakia beach in Syria is being sought and would include protection of a section of sea offshore; however, it is facing strong opposition from the tourism sector (Serra, 2008; Rees et al., 2010).

Marine debris is a significant problem on many green turtle nesting beaches in the eastern Mediterranean, in particular the nesting beaches of Cyprus and Turkey (Camiñas, 2004; Demetropoulos and Hadjichristophorou, 2010; Fuller et al., 2010b; Türkozan and Kaska, 2010). Although organized beach clean-ups in recent years on some beaches in Cyprus have greatly reduced the amount of litter on the beach (Demetropoulos and Hadjichristophorou, 2010; Fuller et al., 2010b), it is still an overall pervasive problem.

6.2.6.1. National Legislation and Protection

In addition to the international mechanisms, most Mediterranean countries have developed legislation to protect sea turtles and/or nesting habitats. They are summarized below. However, the overall effectiveness of this country specific legislation is unknown at this time.

Albania

There are no specific national laws protecting or prohibiting take of sea turtles in Albania (Haxhiu, 2010).

Croatia

The green turtle has been protected under the Nature Protection Act since 1995 by virtue of its inclusion on the Croatia red list of threatened taxa (Lazar, 2010). The Nature Protection Act (Official Gazette 70/05 and 139/08) was last modified in 2008 and is available at http://faolex.fao.org/docs/texts/cro49067E.doc.

Cyprus

Since 1971, sea turtles and their eggs have been protected by Cyprus law (regulations made 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. In addition, two nesting beaches (Lara and Toxeftra) have been afforded protection through the Fisheries Regulation, with a maritime zone extending to the 20 misobath(Margaritoulis, 2007).
Egypt

Minister of Agriculture Decree 1403 of 1990 affords protection to 14 reptile species, including the green turtle (Laurent et al., 1996; Nada and Casale, 2010). The decree prohibits the capture and killing of these species, as well as possessing or selling these species, whether alive or dead, unless permits have been granted for scientific or tourist purposes. An English translation of this Ministerial Decree is available at http://faolex.fao.org/docs/pdf/egy54096E.pdf. Other national laws also aimed at protecting wildlife, including sea turtles, include Law 53 of 1966 that includes 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 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).

Greece

Green 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 the green turtle (Margaritoulis and Panagopoulou, 2010).

Israel

National laws and regulations assist in protecting green turtles in Israel. The National Parks, Nature Reserves, National Sites and Memorial Sites Law of 1998 identifies marine protected areas used by green turtles; the Wildlife Protection Law of 1955 prohibits the hunting of protected wild animals unless special permission is granted; and sea turtle fishing restrictions imposed in 1963 (Levy, 2010).

Italy

Green turtles have been legally protected in Italy since 1981 under the Ministerial Decree of the Ministry of Maritime Affairs of May 21, 1980. They are also protected under the Decree of the Maritime Affairs of May 5, 1989, which includes regulates relative to the taking of several marine species, including the green turtle; and Law 381 of 1988 containing amendments to Law 963 of 1965 on fisheries, which prohibit capture of protected marine species, including green turtles (Casale, 2010).
**Lebanon**

National legislation protecting green turtles in Lebanon includes Ministerial Decision 125 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).

**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).

**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 155 dated June 10, 1987, requests that regional delegates to ensure sea turtle fishing is prohibited (Bradai and Jribi, 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). Three beaches (Belek, Göksu Delta, and
Several regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the Mediterranean Sea. The international instruments listed below apply to sea turtles found in the Mediterranean Sea and are described in Appendix 5.

- Convention on Biological Diversity
- Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention)
- Convention on the Conservation of European Wildlife and Natural Habitats
- Convention on the Conservation of Migratory Species of Wild Animals
- Convention on International Trade in Endangered Species of Wild Fauna and Flora
- Fishery and Agricultural Organization Technical Consultation on Sea Turtle-Fishery Interactions
- International Convention for the Prevention of Pollution from Ships (MARPOL)
- International Union for Conservation of Nature
- Protocol Concerning Specially Protected Areas and Biological Diversity in the Mediterranean
- Ramsar Convention on Wetlands
- United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing
- United States Magnuson-Stevens Fishery Conservation and Management Act

6.3. Assessment of Significant Portion of its Range (SPR)

The SRT reviewed the information on threats and extinction risk within each DPS to determine whether there were portions of the range of the DPS that might be considered “significant portions of the range” in accordance with the definition of endangered and threatened species and the draft policy that interprets that term (see Section 3.5 for more details on the SPR deliberative process).

The extinction risk is relatively uniformly high throughout the range of the DPS, and the threats to the nesting sites within this DPS are relatively uniform in distribution and impact (e.g., bycatch in coastal fisheries gear and harvest in coastal waters of the southern Mediterranean), likely owing to the extremely limited spatial distribution of animals within this DPS. Because the status of rookeries and the nature and degree of threats are relatively uniform across the range of the Mediterranean DPS, the SRT concluded that the need to consider a significant portion of the range does not apply to this DPS.
6.4. Assessment of Extinction Risk

For the SRT’s assessment of extinction risk for green turtles in the Mediterranean DPS, there were two separate sets of ranking exercises: One focusing on the importance that each SRT member placed on each of the six different elements for this region (Table 6.3), and a second which reflects the SRT members’ expert opinion about the probability that green turtles would fall into any one of various extinction probability ranges (Table 6.4). See Section 3.3. for details on the six elements and the voting process.

Table 6.3. Summary of ranks that reflect the importance placed by each SRT member on the critical assessment elements considered for the Mediterranean DPS. For Elements 1-4, higher ranks indicate higher risk factors.

<table>
<thead>
<tr>
<th>Critical Assessment Elements</th>
<th>Element 1</th>
<th>Element 2</th>
<th>Element 3</th>
<th>Element 4</th>
<th>Element 5</th>
<th>Element 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abundance (1 to 5)</td>
<td>3.9</td>
<td>2.7</td>
<td>3.6</td>
<td>3.1</td>
<td>-1.2</td>
<td>0.5</td>
</tr>
<tr>
<td>Trends / Productivity (1 to 5)</td>
<td>0.2</td>
<td>0.2</td>
<td>0.2</td>
<td>0.3</td>
<td>0.2</td>
<td>0.1</td>
</tr>
<tr>
<td>Spatial Structure (1 to 5)</td>
<td>3-5</td>
<td>2–4</td>
<td>3–5</td>
<td>1–4</td>
<td>(-2)–0</td>
<td>0–1</td>
</tr>
<tr>
<td>Diversity / Resilience (1 to 5)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Five-Factor Analyses (-2 to 0)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Conservation Efforts (0 to 2)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>MEAN RANK</td>
<td>3.9</td>
<td>2.7</td>
<td>3.6</td>
<td>3.1</td>
<td>-1.2</td>
<td>0.5</td>
</tr>
<tr>
<td>SEM</td>
<td>0.2</td>
<td>0.2</td>
<td>0.2</td>
<td>0.3</td>
<td>0.2</td>
<td>0.1</td>
</tr>
<tr>
<td>RANGE</td>
<td>3–5</td>
<td>2–4</td>
<td>3–5</td>
<td>1–4</td>
<td>(-2)–0</td>
<td>0–1</td>
</tr>
</tbody>
</table>

With respect to the important rankings for the six elements, the first four elements using the 1-5 ranking system (higher rank equals higher risk factor), nesting abundance featured most prominently in the risk threshold voting, likely owing to the overall small population size in the Mediterranean. Spatial structure (i.e., limited overall nesting distribution) also featured relatively prominently (3.6) in the risk threshold voting.

SRT members also generally thought that future threats not yet reflected in nester abundance or not yet experienced by the population weighed heavier in their risk assessment voting than did any conservation efforts that may emerge in the future. With respect to the diversity of opinions among the SRT members when considering the six Critical Elements, the largest range in rankings (i.e. voter opinion) was noted for Diversity / Resilience Section (w/ ranks from 1 to 4).
Table 6.4. Summary of Green Turtle SRT member expert opinion about the probability that the Mediterranean DPS will reach a critical risk threshold within 100 years. Each SRT member assigned 100 points across the rank categories. The continuum in Row 1 has categories with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability of Reaching Critical Risk Threshold</th>
<th>Mean Assigned Points</th>
<th>SEM</th>
<th>Min</th>
<th>Max</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt;1%</td>
<td>10.1</td>
<td>4.9</td>
<td>0</td>
<td>50</td>
</tr>
<tr>
<td>1%–5%</td>
<td>11.7</td>
<td>3.8</td>
<td>0</td>
<td>30</td>
</tr>
<tr>
<td>6%–10%</td>
<td>17.6</td>
<td>4.2</td>
<td>0</td>
<td>50</td>
</tr>
<tr>
<td>11%–20%</td>
<td>27.9</td>
<td>7.8</td>
<td>0</td>
<td>80</td>
</tr>
<tr>
<td>21%–50%</td>
<td>23.9</td>
<td>6.4</td>
<td>0</td>
<td>80</td>
</tr>
<tr>
<td>&gt;50%</td>
<td>8.7</td>
<td>4.2</td>
<td>0</td>
<td>50</td>
</tr>
</tbody>
</table>

Of the critical risk threshold categories describing the probability that the Mediterranean DPS will reach a critical risk threshold within 100 years (Table 6.4), the SRT member votes resulted in the greatest point (i.e., probability) designations in the '11%–20%' and '21%–50%' risk ranges (mean of 27.9 and 23.9 points, respectively). The '>50%' and '<1%' ranges received the fewest points from SRT members (mean of 8.7 and 10.1, respectively).

In their vote justifications, most members cited the low abundance, limited nesting range, poor level of genetic diversity, and overall high threats as the primary factors that influenced their votes. Additional factors that were cited included the PVA result for West Coast Cyprus, the political instability in the region, the IUCN listing history of Mediterranean green turtles, the lack of information about foraging distribution, and the modest conservation efforts. In general, the vote justifications provided for this DPS were relatively consistent across SRT members, perhaps owing to the fact that the DPS range is the smallest of all DPSs and the fact that the threats and population parameters are relatively consistent throughout.

6.5. Synthesis and Integration

During the consideration of the Mediterranean DPS’s status, an integrated approach was taken by the SRT to consider the many critical assessment elements described earlier. The Mediterranean DPS is characterized by relatively low green turtle nesting abundance, with nesting reported in 32 different locations. There is little if any interchange of individuals present with the Atlantic Ocean. The SRT acknowledged that the low nesting abundance of this DPS creates an intrinsically great risk to the long-term stability of the population.

Population trends, Spatial Structure and Diversity/Resilience in the Mediterranean DPS were considered by the SRT to contribute to the likelihood of extinction of the DPS in the next 100 years. Nesting data gathered in Turkey, Cyprus, and Israel showed no apparent decreasing trend while the PVA for the West Coast, Cyprus estimated a 33.3 percent probability that this population has a 50 percent decline at the end of 100 years.
Coastal development, beachfront lighting, erosion resulting from sand extraction, fishing practices, and marine pollution both at nesting beaches and important foraging grounds is a continuing concern across the DPS. Current illegal harvest of green turtles for human consumption continues as a moderate threat to this DPS. Fishery bycatch occurs throughout the Mediterranean Sea, particularly bycatch mortality of green turtles from pelagic longline, set net, and trawl fisheries, and is a continued threat to this DPS. Additional threats from boat strikes, which are becoming more common, and changes likely to result from climate change will negatively impact this DPS. The SRT considered these threats heavily in the overall critical risk threshold.

The SRT determined the likelihood of reaching a critical risk threshold of extinction within 100 years was relatively high (60.6 percent of votes cast for the ‘>11%’ likelihood categories).
7. SOUTH ATLANTIC DPS (DPS #3)

7.1. DPS Range and Nesting Distribution

The South Atlantic DPS boundary begins between the Panama and Columbian border up to the U.S Virgin Islands and the British Virgin Islands west to Senegal in Africa. Green turtle nesting occurs on beaches throughout the South Atlantic, in the Caribbean portion of the South Atlantic including Caribbean South America, along eastern South America down through Brazil, along the western coast of Africa from southern Mauritania to South Africa, and in the middle of the South Atlantic on Ascension Island (Figure 7.1).

Figure 7.1. Nesting distribution of green turtles in the South Atlantic (blue-shaded area). Size of circles indicates nesting abundance category. Locations marked with ’×’ indicate nesting sites lacking abundance information.

The South Atlantic DPS nesting sites can be roughly divided into four regions: South Atlantic Caribbean (including Colombia, the Guianas, and Aves Island in addition to the numerous small, insular nesting sites), Brazil, Ascension Island, and western Africa. The primary nesting sites for green turtles in the Caribbean South Atlantic are: Aves Island, Venezuela; Galibi and Matapica Reserves, Suriname; and unquantified but substantial nesting in French Guiana. Additional low levels of nesting occur throughout many of the Caribbean Islands in the DPS as well as along Colombia’s Caribbean coast. Further south in the western South Atlantic there are important rookeries off Brazil, on Isla Trindade, Atol das Rocos, and Fernando de Noronha, with smaller rookeries also occurring on the Brazilian mainland coast. Ascension Island, UK is the only green turtle nesting site in the central South Atlantic. In the eastern South Atlantic, primary
green turtles nesting beaches are found along the west coast of the African continent including Bioko Island, Equitorial Guinea; the Bijagos Archipelago, Guinea Bissau; and São Tome and Principe, with scattered, limited nesting on other insular and mainland beaches.

The in-water range of the South Atlantic DPS is similarly widespread. Juvenile and adult green turtles utilize foraging areas throughout the Caribbean areas of the South Atlantic, often resulting in interactions with fisheries occurring in those same waters (Dow et al., 2007). Juvenile green turtles from multiple rookeries also frequently utilize the nearshore waters off Brazil as foraging grounds as evidenced from the frequent captures by fisheries (Marcovaldi et al., 2009; Lima et al., 2010; López-Barrera et al., 2012). Genetic analysis of green turtles on the foraging grounds off Ubatuba and Almofala, Brazil show mixed stocks coming primarily from Ascension, Suriname and Trindade as a secondary source, but also Aves, Suriname, Trindade, and even sometimes Costa Rica (North Atlantic DPS) (Naro-Maciel et al., 2007; Naro-Maciel et al., 2012). While no nesting occurs as far south as Uruguay and Argentina, both have important foraging grounds for South Atlantic green turtles (López-Mendilaharsu et al. 2006, Lezama, 2009; González Carman et al., 2011; Prosdocimi et al., 2012; Rivas-Zinno, 2012). In the eastern South Atlantic, significant sea turtle habitats have been identified, including green turtle feeding grounds in Banc d’Arguin, Mauritania (Fretey, 2001); Corisco Bay, Equatorial Guinea/Gabon (Formia, 1999); Congo; Mussulo Bay, Angola (Carr and Carr, 1991); as well as Principe Island.

7.2. Critical Assessment Elements

In the evaluation of extinction risk for green turtles in the South Atlantic DPS, the SRT considered six distinct elements, each of which are discussed below: (1) Nesting Abundance, (2) Population Trends (3) Spatial Structure, (4) Diversity / Resilience, (5) Five-Factor Threat Analyses, and (6) Existing Conservation Efforts. See Section 3.3 for additional information on the selection of these six Critical Assessment Elements.

7.2.1. Nesting Abundance

For the South Atlantic DPS, we identified 51 total nesting sites. Of those sites, some are individual beaches while others may be multiple nesting beaches lumped together, typically when there is limited nesting and limited data (for example the Caribbean coast of Colombia, mainland Brazil and Venezuela, and most of the Caribbean islands that fall within the South Atlantic DPS nesting area. Much of the South Atlantic is data poor with only occasional or incomplete nesting surveys, and therefore for 37 of the 51 identified nesting areas we were not able to estimate female abundance, even for relatively large rookeries such as French Guiana. The sites for which abundance could not be estimated are Anguilla; Antigua and Barbuda (numerous beaches); Aruba; Barbados; Bonaire; British Virgin Islands; Cape Verde; Colombia (rest of mainland other than Rio Cedros and Monitos); Rio Cedros (Colombia); Monitos (Colombia); Curacao; Dominica; Awala Yalimpo (French Guiana); Pointe Isere, Fazez, Irakumpapi, Organabo (French Guiana); Kourou and Karouaba beaches (French Guiana); Cayenne-Montjoly (French Guiana); Guadalupe; Petite Terre-Terre de Bas (Guadalupe); Petite Terre-Terre de Haut (Guadalupe); Les Galets de Marie-Galante (Guadalupe); Guyana (Luri, Almond, and Tiger beaches); Martinique; Montserrat; St. Kitts and Nevis; St. Lucia; St. Vincent and the Grenadines; Zeelandia Beach (St. Eustatius); St. Maarten; Trinidad; Tobago; Venezuela;
Arembepe, Praia do Forte, Costa do Sauípe, Sitio do Conde, and other scattered nesting (Brazil); St. Croix (USVI); St. Thomas-St. Johns (USVI); Corisco Bay (mainland Equatorial Guinea); Bijagos Archipelago (multiple island sites other than Orango and Poilão; Guinea Bissau).

Of the nesting sites for which an estimate could be derived, Poilão, Ascension Island, and the Galibi Reserve accounted for the bulk of the nesting (Table 7.1). Among the nesting sites with adult female estimates, the largest nesting site, Poilão (in the Bijagos Archipelago, Guinea-Bissau) accounts for almost 46 percent of the total females (Table 7.2). However due to lack of nesting site-specific data the Poilão female abundance estimate was derived using a standard of 3 years for the remigration interval and 3 clutches per female per nesting season. Annual nest numbers on Ascension rival that on Poilão but the estimate of adult female abundance is substantially lower primarily due to using the observed clutch frequency of 6 nests per female per nesting season.

**Table 7.1. Summary of green turtle nesting sites in the South Atlantic.** Data are organized by country, nesting site, monitoring period, and estimated total nester abundance [(Total Counted Females/Years of Monitoring) x Remigration Interval], and represents only those sites with sufficient data to estimate number of females. Many nesting sites, including relatively large ones in the South Atlantic, are data deficient and estimates could not be made for those beaches. For a list of references on abundance data, see Appendix 2.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YEARS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>United Kingdom</td>
<td>Ascension Island</td>
<td>2010–2012</td>
<td>13,417</td>
</tr>
<tr>
<td>Brazil</td>
<td>Atol das Rocas</td>
<td>2005–2008</td>
<td>775</td>
</tr>
<tr>
<td>Brazil</td>
<td>Fernando de Noronha</td>
<td>2008–2013</td>
<td>358</td>
</tr>
<tr>
<td>Brazil</td>
<td>Isla Trindade Island</td>
<td>2008–2010</td>
<td>2,016</td>
</tr>
<tr>
<td>Venezuela</td>
<td>Aves Island</td>
<td>2010</td>
<td>2,833</td>
</tr>
<tr>
<td>Suriname</td>
<td>Matapica Reserve</td>
<td>2008–2010</td>
<td>3,661</td>
</tr>
<tr>
<td>Suriname</td>
<td>Galibi Reserve</td>
<td>2008–2010</td>
<td>9,406</td>
</tr>
<tr>
<td>United States (USVI)</td>
<td>Buck Island</td>
<td>2006–2007</td>
<td>63</td>
</tr>
<tr>
<td>São Tomé and Principe</td>
<td>Principe</td>
<td>2009</td>
<td>76</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>João Vieira</td>
<td>2011</td>
<td>596</td>
</tr>
<tr>
<td>Guinea-Bissau</td>
<td>Poilão</td>
<td>2007</td>
<td>29,016*</td>
</tr>
</tbody>
</table>

Comment [A4]: Comment: The number of nests in Atol das Rocas and Noronha are very different, while Atol das Rocas is in the upper part of the bin (100-500), Noronha is in the lower part or even less than 100. Here it is the exact number of nests of the past seasons:
2008- 55
2009- 55
2010- 190
2011- 76
2012- 134
As a reference you can use SWOT (2008-2012) Using the same parameters as those from Rocas (bellini et al. 2013), and the mean number of nests between 2008 and 2012 (102 nests) Nesters/yr = 102/5.2 = 20
Total number of esters = 20*3.5 = 70
Thus for example, nester abundance should be around 70

Comment [A5]: Response: Made the suggested changes. The VTP table questioned the validity of the data found for this nesting site. So this is more updated and valid.

Comment [A6]: Response: This island has been referred to as Isla Trindade in other places in the document.
* Estimated females may be biased high. Past data has not shown RI and clutch frequency data specific to nesting site. Used average clutch frequency of three (3) for each.

Table 7.2. Green turtle nester abundance distribution among nesting sites in the South Atlantic.

<table>
<thead>
<tr>
<th>NESTER ABUNDANCE</th>
<th># NESTING SITES DPS 3</th>
</tr>
</thead>
<tbody>
<tr>
<td>n/a</td>
<td>37*</td>
</tr>
<tr>
<td>1–10</td>
<td>0</td>
</tr>
<tr>
<td>11–50</td>
<td>0</td>
</tr>
<tr>
<td>51–100</td>
<td>2</td>
</tr>
<tr>
<td>101–500</td>
<td>3</td>
</tr>
<tr>
<td>501–1000</td>
<td>3</td>
</tr>
<tr>
<td>1001–5000</td>
<td>3</td>
</tr>
<tr>
<td>5001–10000</td>
<td>1</td>
</tr>
<tr>
<td>10001–100000</td>
<td>2**</td>
</tr>
<tr>
<td>&gt;100,000</td>
<td>0</td>
</tr>
</tbody>
</table>

TOTAL SITES 51
TOTAL ABUNDANCE 63,332
PERCENTAGE at LARGEST NESTING SITE 45.8% (Poilão, Guinea-Bissau)

*There are issues with lack of data, even at some of the relatively large rookeries such as in French Guiana, which likely lowers the nester abundance estimate (37 of 51 rookeries have insufficient data to estimate abundance).

**There is some question about the estimated size of the largest nesting site (Poilão) due to data uncertainty.

15.1.1. Population Trends

Despite the numerous and widespread nesting beaches in this DPS, long-term monitoring data is relatively scarce. There are only three sites for which 10 or more years of recent data are available for annual nester abundance (the standards for representing trends in bar plot in this report; Figure 7.2). Of these, no sites met our standards for conducting a PVA (see Section 3.2 for more on data quantity and quality standards used for bar plots and PVAs). While trends cannot be estimated in many cases due to the lack of data, we discuss the indications of possible trends at some of the primary nesting sites. For a list of references on trend data, see Appendix 3.
Abundance data for green turtle nesting in the South Atlantic DPS with greater than 10 years of recent monitoring data, although with some missed years. These sites are Ascension Island (12 yrs, not counting the 1822 monitoring), Atol das Rocas, Brazil (17 yrs), and Galibi Reserve and Matapica (2 combined sites; 33 yrs). Note that numbers for Ascension Island are presented as number of nesters, whereas Atol das Rocas and Galibi Reserve and Matapica are presented as number of nests.

The only nesting concentration in the central Atlantic, and one of the largest in the South Atlantic DPS, is at Ascension Island (United Kingdom). This population has increased substantially over the last three decades (Broderick et al., 2006; Glen et al., 2006b). Mortimer and Carr (1987) counted 5,257 nests in 1977 (about 1,500 females), and 10,764 nests in 1978 (about 3,000 females) whereas from 1999–2004, a total of about 3,500 females nested each year (Broderick et al., 2006). In 2012, radio transmitters were deployed on 40 turtles to easily locate them each time they nested. It was found that on average, each female lays 6 clutches of 120–150 eggs per season—double the previous estimate (Weber et al., 2013). Since 1977, numbers of nests on one of the two major nesting beaches, Long Beach, have increased exponentially from around 1,000 to almost 10,000 (http://www.ascension-island.gov.ac/government/conservation/our-species/marine-turtles/). From 2010 to 2012, an average of 23,000 nests per year was laid on Ascension (S. Weber, Ascension Island).
Government, pers. comm., 2013). These data are suggestive of an increase, although historic data from additional years are needed to fully substantiate this possibility.

Off northern Brazil, nesting on Atol das Rocas has high annual variability but appears to be overall stable from the 1990s through 2008 (from abundance data in Bellini et al., 2012). The southernmost nesting concentration in the Western Atlantic is at Isla Trindade, Brazil. This nesting population has been stable with a mean of ca. 1500–2000 females nesting per year since the early 1980s (Moreira et al., 1995; Moreira and Bjorndal, 2006; Almeida et al., 2011a). In 2010, more than 22,175 clutches were recorded (A. Turny, WWF, pers. comm., 2013), which translates to approximately 7,600 females (assuming 3 clutches per female). In Fernando de Noronha, despite no data have been published yet, nesting numbers are increasing; the average on the first decade of monitoring was 30 nests per year (from 1984 to 1993) and the average in the last decade (from 2002 to 2013) is 90 nests per year (unpublished data, Projeto Tamar Database /A. J. B. Santos, TAMAR, pers. comm., 2014).

The nesting concentration at Galibi Reserve and Matapica in Suriname was stable from the 1970s through the 1980s, albeit at a reduced level following extensive egg harvest in the 1960s. From 1975–1979, 1,657 females were counted (Schulz, 1982), a number that increased to a mean of 1,740 females from 1983–1987 (Mohadin and Ogren, 1989), and to 1,803 females in 1995 (Weijerman et al., 1998). Since 2000, there appears to be a rapid increase in nest numbers.

At Aves Island, Venezuela, the population appears stable to slightly increasing. From 1984–1987, 700–900 nests (about 230–300 females) per season were counted; in 1997, a total of 267 females nested based on number of nests seen and a clutch frequency of 3.0 nests/season (V. Vera, Dirección General de Fauna, pers. comm. to K. Eckert, WIDECAST, 2001); and in 2005 and 2006, a total of 335 and 443 females nested, respectively (Vera and Montilla, 2006; Vera, 2008). In 2008, an estimated 669 females nested (Vera and Buitrago, 2012).

There are two areas of interest in the eastern portion of the South Atlantic Ocean: Bioko Island (Equatorial Guinea) and the Bijagos Archipelago (Guinea-Bissau). Nesting at Bioko Island appears to have decreased, whereas nesting in the Bijagos Archipelago may be stable; however, the lack of long-term and/or multiple year data sets preclude meaningful trend assessment for both sites. At Bioko, the number of nightly emergences during the peak of the nesting season declined from 200–300 females per night during the 1940s (Eisentraut, 1964) to 50–100 females per night in the 1980s; J. Tomas, University of Valencia-Spain, pers. comm., 2001). During the 1996–1997 and 1997–1998 nesting seasons, a mean of 1,468 nests were deposited (approximately 500 females; Tomás et al., 1999). In 2010, approximately 1,700 nests were deposited. In the Bijagos Archipelago, Parris and Agardy (1993 as cited in Fretey, 2001) reported approximately 2,000 females per season from 1990–1992, and Catry et al., (2002) reported approximately 2,500 females nesting during the 2000 season. Given the typical large annual variability in green turtle nesting, Catry et al. (2009) suggest it is premature to consider there to be a positive trend in Poilão nesting, though others have made such a conclusion (Broderick et al., 2006). Despite the seeming increase in nesting, interviews along the coastal areas of Guinea-Bissau generally resulted in the view that sea turtles overall have decreased noticeably in numbers over the past two decades (Catry et al., 2009). In 2011, a record estimated
50,000 green turtle clutches were laid throughout the Bijagos Archipelago (P. Catry, Instituto Superior de Psicologia Aplicada, pers comm., 2012).

15.1.2. Spatial Structure

When examining spatial structure for the South Atlantic DPS, the SRT examined three lines of evidence, including genetic data, flipper and satellite tagging, and demographic data.

While the eastern Caribbean rookeries (St. Croix, Aves Island, and Suriname) are dominated by a shared haplotype and show strong reproductive isolation from other rookeries both in the western Caribbean and from Brazilian rookeries in the South Atlantic, the phylogenetic relationship of the eastern Caribbean rookeries indicate that despite the close proximity of the other Caribbean rookeries, they are more closely related to the rookeries of the South Atlantic (M. Jensen, NRC, unpubl. data). Although the rookeries in the western Caribbean are located in a transition zone between the Caribbean and the Atlantic rookeries with turtles foraging both north, south and west. It seems that the haplotype CM-A8 common among south Atlantic rookeries has only been found in low numbers in foraging populations of juvenile green turtles of the North Atlantic (Bass et al., 2006).

The south Atlantic green turtle rookeries found in Brazil, Ascension Island, and West Africa have shallow structuring and are dominated by a common and widespread haplotype CM-A8 that is found in high frequency across all rookeries (Bjorndal et al., 2006; Formia et al., 2006). This also, results in non-significant FST values from distant rookeries such as Ascension and Bioko and between Principe and Sao Tome.

In the Southwest Atlantic, foraging areas in Brazil are mainly made up of turtles from Ascension Island, Trinidad and to some degree from Suriname (Nardo-Maciel et al., 2007, 2012). Because of the overlap in haplotype frequencies there is insufficient resolution in the genetic data to determine if there is any dispersal across the South Atlantic (Naro-Maciel et al., 2012). Overall, the distribution of the two genetic haplotype lineages (Clade I and Clade II) is very similar to what is seen for the nesting population and indicates a strong regional structuring with little overlap. There is however an overlap in foraging areas between the eastern and western Caribbean rookeries (North Atlantic DPS and South Atlantic DPS). Lahanas et al. (1998) showed that a significant proportion of juvenile green turtles in the Bahamas originate from the eastern Caribbean (Aves Island/Suriname) (12.9 percent).

While loggerhead turtles are known for their trans-Atlantic dispersal, green turtles are generally thought to disperse across smaller distances within their natal regions. A recent study, however, showed that a large proportion of juvenile green turtles in Cape Verde in the eastern Atlantic originated from distant rookeries across the Atlantic, namely Suriname (38 percent), Ascension Island (12 percent) and Guine Bissau (19 percent) suggesting that, like the loggerheads, green turtles in the Atlantic undertake transoceanic developmental migrations (Monzon-Arguello et al., 2010). The fact that long distance dispersal is only seen for juvenile turtles suggests that larger adult-sized turtles return to forage within the region of their natal rookeries thereby limiting the potential for gene-flow across larger scales (Monzon-Arguello et al., 2010). It is very likely that juvenile turtles from western African rookeries make similar trans-oceanic migrations to the
coast of Brazil but the genetic similarity across this region makes this difficult to definitively conclude. Satellite and flipper tag recoveries (often with accompanying genetic analysis described above) further show the wide range of the DPS and the interconnectedness of the different regions via juvenile migrations to foraging grounds both near to and far from the natal beaches. Movement between feeding grounds and rookeries in the Caribbean (N. Atlantic DPS) and Brazil (Ceará) (S. Atlantic DPS) has been established by flipper tag recoveries (Godley et al., 2003; Lima et al., 2003; 2008). Ubatuba and Almofala, important juvenile green turtle foraging grounds off Brazil are utilized by mixed stocks. They come mostly from Ascension (Naro-Maciel et al., 2012), but also Suriname, Aves, and Trindade (Brazil). At Almofala, Costa Rican greens are also present. (Naro-Maciel et al., 2007).

Movement between feeding grounds and rookeries in the Caribbean (N. Atlantic DPS, e.g. Suriname, Costa Rica and Puerto Rico) and Brazil (Ceará) (S. Atlantic DPS) has been established by flipper tag recoveries (Godley et al., 2003; Lima et al., 2003; 2008; 2012).

Demographic data are limited and inconsistent for many nesting sites in this DPS (see Section 2). Overall a variety of demographic parameters of green turtles in the South Atlantic appear to vary widely among the various nesting assemblages, for a variety of demographic parameters. This variability in parameters such as remigration interval, clutch size, hatching success, sex ratio and clutch frequency is not separated out regionally within the DPS and, therefore, does not necessarily suggest a high level of population structuring. Hatching success varies widely from 54-94 percent for areas with available information. Most of the data were collected in the 1970’s to the mid 1980’s. Clutch size range varies widely (102–138 eggs/nest) with an approximate midpoint of 120 eggs/nest. Clutch frequency ranges from 1.6 to 6, with 3 as an approximate midpoint. Remigration interval varies from 2.3 years to 3.5 years by nesting site. The estimated age to maturity is 17–35 years (Frazer and Ladner, 1986). In a comparison of average nester sizes Hirth (1997a) determined that nesters at Isla Trindade, Brazil (average CCL 116.8 cm), Atol das Rocas, Brazil (average CCL 118.6 cm), and Ascension Island (average CCL 116.8 cm) are among the largest nester sizes reported for green turtles globally. Other studies in those areas have found somewhat smaller average sizes than those reported in Hirth (1997a), but still larger than most other geographical areas: 115.2 cm average CCL at Isla Trindade (Almeida et al., 2011) and 115.9 cm CCL (1990–1992) down to 112.9 cm CCL (2006–2008) for Atol das Rocas (Bellini et al., 2013). The nester sizes were as follows: mean CCL=116.8 cm at Isla Trindade, Brazil; mean CCL=118.6 cm Atol das Rocas, Brazil; and mean CCL=116.8 cm at Ascension Island. These are among the largest reported for green turtle populations worldwide (Hirth, 1997).

15.1.3. Diversity / Resilience

The components considered under this critical assessment element include the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation (e.g., high vs. low beach face, insular vs. continental nesting sites), and the genetic diversity within the DPS. Components such as these are important considerations for assessing the potential impact of catastrophic events such as storms, sea level rise, and disease.
As presented above, the overall range of the DPS is extensive and varied. Nesting is widespread throughout the DPS, with multiple rookeries in the South/Southeastern Caribbean and Caribbean South America, Brazil (particularly Brazilian islands), western Africa (primarily Poilão and Bioko, but also on Sao Tome, other islands, and some scattered nesting on the mainland), and Ascension Island. Ascension Island, one of the largest rookeries, is isolated and protected in the middle of the South Atlantic, and appears to have migratory connections to rookeries on the eastern and western ends of the DPS. The insular sites vary quite a bit in terms of potential impacts from sea level rise and tropical weather. Aves Island, one of the largest Caribbean rookeries in the SA DPS is particularly vulnerable to sea level rise as it is a very low-lying island.

The nesting sites found in Brazil, Ascension Island and western Africa have shallow structuring and are all dominated by a common shared haplotype found in high frequency across those nesting beaches. Meanwhile the eastern Caribbean rookeries (primarily St. Croix, Aves Island, and Suriname) are dominated by another shared haplotype and appear to be largely reproductively isolated from other rookeries in the western Caribbean and Brazil. However, despite the geographic closeness to the remainder of the Caribbean (which falls in the North Atlantic DPS), the eastern Caribbean nesting sites are more closely related to the other nesting sites of the South Atlantic DPS.

Individuals from one of the largest nesting sites, Ascension Island, must migrate long distances to reach foraging grounds as little forage is available near the island. The foraging grounds off Brazil, Uruguay, and Argentina appear to be of primary importance for many of the juveniles and adults originating from the western nesting sites in the DPS and for Ascension nesters. Individuals from the Caribbean portion of the South Atlantic DPS appear to forage off Brazil as well as at seagrass beds off Central America in what is the North Atlantic DPS. The Gulf of Guinea is an important foraging ground for individuals originating in western Africa nesting sites, but juveniles from Suriname and Ascension are also relatively common in African foraging grounds.

15.1.4. Analysis of Factors Listed Under ESA Section 4(a)(1)

Section 4(a)(1) of the ESA lists five factors (threats) that must be considered when determining the status of a species. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

The following information on these factors / threats pertains to all green turtles that may be found in the South Atlantic DPS. Because it is possible that oceanic juveniles from this DPS also are found in the North Atlantic DPS, especially parts of the Caribbean outside of the South Atlantic DPS area, the narratives for those regions should also be consulted. Likewise, some foraging areas in the South Atlantic are used by individuals from other DPSs. For example,
while most of the individuals foraging in waters off Almofala, Brazil come from South Atlantic nesting grounds (Ascension, Aves, etc.) individuals from other DPSs may forage in those areas as well, such as turtles from Costa Rica (North Atlantic DPS; Naro-Maciel et al., 2007). See Appendix 3 for a summary of threats by DPS.

15.1.4.1. Factor A: Destruction or Modification of Habitat or Range

Coastal development, beachfront lighting, fishing practices, marine pollution, sea level rise, and erosion persist as threats to this DPS.

Terrestrial Zone

As the main nesting sites for green turtles in Brazil are located in the oceanic islands which are MPA, females, eggs and hatchlings are fully protected (Almeida et al., 2011a; Bellini et al., 2013). As for the fewer continental records, destruction and modification of sea turtle, including green turtle, nesting habitat in the South Atlantic DPS result from coastal development and construction, placement of erosion control structures and other barriers to nesting, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach sand placement, beach pollution, removal of native vegetation, and planting of non-native vegetation (D’Amato and Marczewski, 1993; Marcovaldi and dei Marcovaldi, 1999; Naro-Maciel et al., 1999; Marcovaldi et al., 2002).

The portion of the Wider Caribbean that falls within the South Atlantic DPS includes what has been called the Eastern Caribbean, Southern Caribbean, and Guianan ecoregions, as well as the Colombian coast within the Southwestern Caribbean ecoregion (Spalding et al., 2007). Throughout the Caribbean ecoregions within the South Atlantic DPS, green turtle nesting is widespread but generally occurs in relatively low numbers at each nesting location, with only Aves Island, French Guiana, and Suriname reporting more than 500 crawls per year, while the majority of the nesting sites have fewer than 25 crawls per year (Dow et al., 2007). Because of the many islands and coastal nesting sites in the Caribbean that falls within the South Atlantic DPS (22 nations and territories, some with multiple nest sites) there is substantial variation in what terrestrial zone habitat issues exist from site to site, but across the area most of the habitat issues cited above apply to some degree. For a thorough breakdown of each nesting area please see Dow et al. (2007) and the individual country reports included as an appendix in the online digital version of the report (http://ufdc.ufl.edu/AA00000379/00001/). In Suriname, nesting beaches tend to shift over time due to a natural cycle of erosion and accretion that occurs by siltation from rivers, wave action, and currents. As a result, coastal development is not a major problem in Suriname (Reichart and Fretey, 1993).

Green turtle nesting in Brazil occurs primarily on oceanic islands, with Trindade Island being the largest nesting site in Brazil. At around 3,600 nests per year, Trindade is also the seventh largest green turtle nesting colony in the Atlantic, and the fourth largest in the South Atlantic [Almeida et al., 2011a]. Atol das Rocas, off northeastern Brazil, is another significant nesting site, and is located within the Atol das Rocas Biological Reserve that incorporates the atoll and surrounding waters was established in 1979 and provides near complete protection for nesting sea turtles on
Erosion is a problem along the long stretches of high energy ocean shoreline of Atlantic Africa and is further exacerbated by sand mining and harbor building (Formia et al., 2003); crumbling buildings claimed by the sea may present obstructions to nesting females. Such nesting beach habitat loss is a concern in Ghana where a combination of sand mining, development, and heavy erosion is occurring. In one stretch of coastline at Beyin, over 600 m along the 10 km beach has been rendered unsuitable for sea turtle nesting as a result (Tanner, 2013). Garbage also litters many developed beaches (Formia et al., 2003). Additional conservation challenges are expected for Bioko Island as well. While somewhat isolated from development in the past, oil resource development in the Gulf of Guinea has driven economic development in the area, and new roads are now planned, which are expected to result in greater access and more development and activities along the beach (Fitzgerald et al., 2011). The Bijagós Archipelago, Guinea-Bissau, especially the island of Poilão, represents one of the most significant nesting colonies in the Atlantic, and the largest green turtle nesting colony along the western coast of the African continent. An estimated 7,397 clutches were laid in 2000 (Catry et al., 2002), with just over 29,000 nests estimated in 2007 (Catry et al., 2009). Given the typical large annual variability in green turtle nesting, Catry et al. (2009) feels it is premature to consider there to be a positive trend in Poilão nesting, though others have made such a conclusion (Broderick et al., 2006). Despite the seeming increase in nesting, interviews along the coastal areas of Guinea-Bissau generally resulted in the view that sea turtles overall have decreased noticeably in numbers over the past two decades (Catry et al., 2009). While some nesting occurs along the African coast near the northern extent of the South Atlantic DPS in Senegal it is thought that nesting was likely more common in the past (Fretet, 2001).

One of the largest nesting colonies in the South Atlantic, at over 11,000 nests/yr, occurs on Ascension Island (Broderick et al., 2006; Almeida et al., 2011). This colony used to be a major source of turtle harvest, but since the 1970’s annual nesting has increased by ca. 28.5 percent, although it still remains below 50 percent of its carrying capacity (Broderick et al., 2006). Threats to green turtle nesting habitat on Ascension Island include mining of beach sand, light pollution, the potential for fuel spills from tankers and on-land storage facilities, litter/debris, invasive vegetation, sea level rise and erosion (Broderick et al., 2002b).

In very low-lying islands such as Aves, rising sea levels and increased storms could result in a loss of nesting habitat; thus potentially eliminating their functionality as nesting beaches.

**Neritic/Oceanic Zone**

Human activities that affect bottom habitat in the green turtle neritic and oceanic zones include fishing practices, channel dredging, sand extraction, marine pollution, and climate change. General human activities have altered ocean ecosystems, as identified by ecosystem models (http://www.lme.noaa.gov). On the western side of the South Atlantic, the Brazil Current Large Marine Ecosystem (LME) region is characterized by the Global International Waters Assessment.
(GIWA) as suffering severe impacts in the areas of pollution, coastal habitat modification, and overexploitation of fish stocks (Marques et al., 2004). The Patagonian Shelf LME is moderately affected by pollution, habitat modification, and overfishing (Mugetti et al., 2004). In the Canary Current LME, the area is characterized by the GIWA as severely impacted in the area of modification or loss of ecosystems or ecotones and health impacts, but these impacts are decreasing (http://www.lme.noaa.gov). The Celtic-Biscay Shelf LME is affected by alterations to the seabed, agriculture, and sewage (Valdés-González and Ramírez-Bautista, 2002). The Gulf of Guinea has been characterized as severely impacted in the area of solid wastes by the GIWA; this and other pollution indicators are increasing (http://www.lme.noaa.gov). On the eastern side of the South Atlantic, the Benguela Current LME has been characterized as moderately impacted in the area of overfishing, with future conditions expected to worsen by the GIWA (Prochazka et al., 2005). Climate change also may result in future trophic changes, thus impacting green turtle foraging grounds.

Coastal degradation can be of particular concern for green turtles as a result of their reliance on ecologically sensitive seagrass and algae areas. In Brazil, green turtles in degraded coastal areas that have been impacted by plastic debris ingestion have been found to have diets that are lower in diversity and quality than those in lesser impacted areas, potentially impacting growth, development, and fecundity (Santos et al., 2011). Off the northwestern coast of Suriname run-off from rice production and other agricultural activities is a problem (Reichart and Fretey, 1993) and likely would have similar impacts. The reduction of carrying capacity for green turtles in seagrass beds impacted by anchor damage in popular bays in the U.S. Virgin Islands has also been documented (Williams, 1988), and likely occur in other similar areas throughout the South Atlantic. Likewise, sediment contamination from coastal and upstream industrial sites has been recognized in the Caribbean, including St. Croix (Ross and DeLorenzo, 1997), and has the potential to impact green turtle habitat as well as the turtles themselves. Such coastal degradation has been seen throughout the Caribbean areas that fall within the South Atlantic DPS (Dow et al., 2007) and it is likely that similar situations occur throughout the coastal areas of the South Atlantic.

Additionally, fishing is a major source of ecosystem alteration of the neritic and oceanic green turtle habitats in the region due to the removal of great amounts of biomass. Fishing effort off the western African coast is increasing and record low biomass has been recorded for exploited resources, representing a 13 times decline in biomass since 1960 (see Palomares and Pauly, 2004). The Celtic-Biscay Shelf LME, the Iberian Coastal Ecosystem LME, the Canary Current LME, and the Guinea Current LME all are severely overfished, and effort now is turning to a focus on pelagic fisheries, whereas historically there were demersal fisheries. The impacts continue to increase in the Guinea Current LME despite efforts throughout the region to reduce fishing pressure (http://www.lme.noaa.gov). Similarly fishing activities have reached the limits of productivity and impacted the ecosystems through changes in trophic webs along the Brazil-Guianas continental shelf, as well as causing direct alteration of seafloor and other habitats from gear (Charlier et al., 2000). Similar impacts are seen in many other places in the South Atlantic and worldwide.

15.1.4.2. Factor B: Overutilization
Overutilization for commercial purposes likely was a factor that contributed to the historical declines of this DPS. Current legal and illegal collection of eggs and harvest of turtles throughout the South Atlantic DPS for human consumption as described below persists as a threat to this DPS. A summary of the intentional impacts is given below.

**Egg and Turtle Harvest**

Deliberate hunting of green turtles for their meat, shells, and eggs is reduced from previous exploitation levels, but still exists. Limited numbers of eggs are taken for human consumption in Brazil, but the relative amount is considered minor when compared to historical rates of egg collection (Marcovaldi and Marcovaldi, 1999; Marcovaldi et al., 2005; Almeida and Mendes, 2007). Use of sea turtles, including green turtles, for medicinal purposes occasionally occurs in northeastern Brazil (Alvez and Rosa, 2006; Braga-Filho and Schiavetti 2013). As an exception in Ceará there are records of illegal commerce and consumption of green turtle meat (E. Lima pers. comm.) Despite being established as a federal biological reserve in 1979, egg harvesting and the poaching of nesting green turtles on Atol das Rocas occurred up through around 1990 when the island became frequented by regular sea turtle conservation activities, wildlife researchers, and Brazilian environmental officers (Bellini et al., 2013). Extensive harvest of nesting females and eggs was common in Suriname for local consumption and export up through about 1940, with many hundreds to over a thousand adults being slaughtered each year (Reichart and Fretey, 1993). Egg harvest continuing unabated for decades beyond that, reaching levels of as much as 90 percent of all eggs laid in the Galibi area by 1967 until a ban was enacted. Subsequently a limited harvest by the locals was allowed via permit, but poaching remains a problem (Reichart and Fretey, 1993).

Throughout the Caribbean areas of the South Atlantic DPS, harvest of green turtle eggs and turtles, both illegal and legal continues (Dow et al., 2007). Among the British Caribbean territories within the South Atlantic DPS (including Anguilla, Turks and Caicos, the British Virgin Islands, and Montserrat) there are legal sea turtle fisheries, with anywhere from a few (Montserrat) to over a thousand (Turks and Caicos) green turtles taken per year (Godley et al., 2004).

Turtles are harvested along the African coast and, in some areas, are considered a significant source of food and income due to the poverty of many residents along the African coast (Formia et al., 2003). On Bioko sea turtle nesting beach protection and monitoring levels are inconsistent and depend on government or other funding for nesting surveys which help deter poaching. After the end of surveys in 1998 around 250 green turtles/year were documented being transported to local markets (Tomás et al., 2010). In the Bijagós Archipelago (Guinea-Bissau) all sea turtles are protected by national law, but enforcement is limited and many turtles are killed by locals for consumption. In 2007, at least 374 adult female green turtles were killed at the Orango National Park (Catry et al., 2009). Targeted captures at sea by foreign fishermen have also been reported (Catry et al., 2009).

15.1.4.3. **Factor C: Disease or Predation**
The primary known disease of significance in the South Atlantic is FP. This disease is highly variable in its presence and severity throughout the area, with areas of lower water quality, especially nutrient enrichment, often being the sites with the most prevalent and most severe cases of FP. In Brazilian waters, FP has been documented but is highly variable among sites. In Fernando de Noronha, a prime green turtle feeding area off northeast Brazil, no FP has been documented. The area is considered pristine, with good water quality. In Espírito Santo Bay, which has experienced substantial environmental degradation, FP was prevalent, occurring on 58.3 percent of individuals, and being found on 41 percent of individuals that had previously been free of FP but were later recaptured in the area (dos Santos et al., 2010). Further south in Brazil, off Santa Catarina and Rio Grande do Sul states, infection rates ranged from zero to 21.4 percent from 1994–2004, with the higher rates primarily occurring in the later years (Rodrigues et al., 2012).

Within the Caribbean, FP has been noted in St. Croix, potentially as early as 1971 based upon records of what at that time was an unidentified disease noted on a green turtle captured in the area (Eliazar et al., 2000). Throughout the 1980s and 1990s there was a noted increase in FP throughout the Caribbean (Williams and Bunkley-Williams, 2000). Similarly in Aves Island, despite monitoring green turtle nesting since 1979, the first case of FP was not documented until 1986 (Sole and Azara, 1996).

FP has been confirmed among green turtles of Africa’s Atlantic coast, from Gabon and Equitorial Guinea (Formia et al., 2007; Girard et al., 2013), Guinea-Bissau (3 cases reported in 2000; Catry et al., 2009), Gambia, and Senegal (Barnett et al., 2004), the Congo and Princeipe Island (Girard et al., 2013). The prevalence varies greatly between locations. A 17 percent prevalence was seen in Corisco Bay from 1998–2006 and a range of 8 percent to 12 percent occurred in Loango Bay and Pointe Indienne, Congo from 2005–2012 (Girard et al., 2013). Meanwhile, an examination of 274 nesting green turtles on Poião, Guinea-Bissau in 2007 did not turn up any clear signs of FP, though a few small, smooth, low growths of unknown cause were seen (Catry et al., 2009). The apparent rarity of FP in turtles of Poião may be a result of the relatively low pollution and the healthy ecosystems at the foraging grounds (Catry et al., 2009), which is similar to what was seen in the Brazilian foraging areas. The contrast of elevated disease rates in areas with poor water quality and high nutrient loading with lower rates of FP in more pristine, low nutrient waters is consistent with the conclusions from Van Houtan et al. (2010).

Depredation of eggs, hatchlings, and adults is also documented within the South Atlantic. Eggs and nests in Brazil experience depredation, primarily by foxes (Vulpes vulpes; Marcovaldi and Laurent, 1996). Nests laid by green turtles in the southern Atlantic African coastline experience predation from local wildlife and feral animals (e.g., jackals; Canis sp.) depredate green turtle nests in Angola (Weir et al., 2007). Shark predation on green turtles, especially by tiger sharks (Galeocerdo cuvier), has been documented off northeastern Brazil at a frequency high enough to indicate that green turtles may be an important food source for tiger sharks off Brazilian waters (Bomatowski et al., 2012). Predation on nesting females can also occur from large predators, such as jaguars (Panthera onca) in Suriname (Autar, 1994). On Ascension Island predation by domestic and feral cats (Felis sp.) and dogs (Canis sp.), frigate birds (Fregata minor), land crabs (subphylum Crustacea), and fish (class Osteichthyes) have all been cited as mortality
sources for hatchling green turtles (Broderick et al., 2002b). Nest predation by introduced roof rats (*Rattus rattus*) was noted as a problem on Buck Island Reef National Monument off St. Croix, but a 1998–2000 program to eradicate the pest species was successful and nest predation by introduced rats has ceased (Witmer et al., 2007). Nest predation by monitor lizards (*Varanus* sp.) on the Bijagos Archipelago was highly variable, with green turtle nests experiencing 76 percent predation rates during the first 10 days after oviposition on João Vieira, but no evidence of predation on Poilão (da Silva Ferreira, 2012). Predation in some areas can come from a wide variety of species, with ghost crabs (family Ocypodidae), ants (family Formicidae), monitor lizards, monkeys (suborder Haplorrhini), porcupines (order Rodentia), vultures (family Accipitridae) and crows (*Corvus* sp.), in addition to village dogs, all preying on eggs and hatchlings on the southern beaches of Bioko, in the Gulf of Guinea (Tomás et al., 1999).

The proliferation of harmful algal blooms (HABs) worldwide (Gilbert et al., 2005) may also impact green turtles in the South Atlantic.

15.1.4.4. **Factor D: Inadequacy of Existing Regulatory Mechanisms**

Our review of regulatory mechanisms under Factor D demonstrates that although regulatory mechanisms are in place that should address direct and incidental take of South Atlantic green turtles and impacts to their habitats, these regulatory mechanisms are insufficient or are not being implemented effectively to protect green turtles. We find that there is a threat from the inadequacy of existing regulatory mechanisms for fishery bycatch and pollution prevention (Factor E), overutilization from legal and illegal takes (Factor B), especially in the Caribbean and Atlantic Africa, and impacts to nesting beach and foraging habitat (Factor A).

The management of sea turtles is facilitated by a number of regulatory instruments at international, regional, national, and local levels, and nearly all countries within the DPS have some level of national legislation directed at sea turtle protection. There are a minimum of 20 national and international treaties and/or regulatory mechanisms that pertain to the South Atlantic DPS. Hykle (2002) and Tiwari (2002) reviewed the value of some international instruments, which vary in their effectiveness. Often, international treaties 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, 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 a 2002 special issue of the Journal of International Wildlife Law and Policy: International Instruments and Marine Turtle Conservation (Hykle, 2002).

Although conservation efforts to protect some nesting beaches are underway, more widespread and consistent protection is needed. Although national and international governmental and non-governmental entities are currently working toward reducing green turtle bycatch, it is unlikely that this source of mortality can be sufficiently reduced across the range of this DPS in the near future because of the lack of bycatch reduction in commercial and artisanal fisheries operating within the range of this DPS, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical
complexities, limitations on enforcement capacity, and lack of availability of comprehensive bycatch reduction technologies.

15.1.4.5. Factor E: Other Natural or Manmade Factors

The South Atlantic DPS of the green turtle is negatively affected by both natural and anthropogenic impacts as described below in Factor E. Within Factor E, we find that fishery bycatch that occurs throughout the South Atlantic, particularly bycatch mortality of green turtles from nearshore gill net fisheries, continues as a threat to this DPS. In addition, changes likely to result from climate change are also a threat to this DPS.

Incidental Bycatch in Fishing Gear

Incidental capture of sea turtles in artisanal and commercial fisheries is a threat to green turtles in the South Atlantic DPS. Green turtles may be caught in pelagic and demersal longlines, drift and set gill nets, bottom and mid-water trawling, fishing dredges, pound nets and weirs, haul and purse seines, pots and traps, and hook and line gear.

Coastal gill net fisheries may be of particular concern to green turtles in the area as many occur in the turtles’ foraging grounds. There is substantial documentation of the intersection of small-scale artisanal gill net fisheries with green turtles in their foraging grounds along the western South Atlantic. A first assessment of the Brazilian states of São Paulo (coastal gill net and pound nets) and Ceará (corrals) estimated 1,874 interactions (2002–2007) with gill nets, 4,517 (1991–2007) with pound nets, and 670 (1993–2007) with corrals (Marcovaldi et al., 2009). Such interactions have been documented in Paranaguá Bay, Brazil (López-Barrera et al., 2012) where 13 percent of observed fishing events had interactions with juvenile green turtles (most captures coming from gill nets) with 63 percent found dead in the nets. Prior to the 2007 ban in Brazil, set nets used for lobster fishing off the Ceará Coast of Brazil killed tens to hundreds of juvenile green turtles, among other sea turtle species, each year (Lima et al., 2010). In addition, juvenile green turtles are captured in estuarine fishing traps (Nagaoka, 2012). Throughout the coast of Brazil the most common species found stranded is the green turtle in juvenile stage (Barata et al., 2011).

Similarly, artisanal gill net fisheries in the coastal waters of the Río de la Plata area of Uruguay was estimated to have captured 497 juvenile green turtles per year during a 2004–2005 observation period (Lezama, 2009). Of those captured, 38 percent were found dead in the nets, with no estimate of post-release mortality (Lezama, 2009). A subsequent study (Rivas-Zinno, 2012) conducted in the area in 2009–2010 following the implementation of a 2008 time-area closure shows the high degree of variability in green turtle use of the area, as the Catch Per Unit Effort (CPUE) increased dramatically and the author estimated 1861 green turtles captured over the 13 month duration of the study, despite the time-area closure during the “peak” season identified in Lezama (2009). Information gathered during the study indicated that there were unusual oceanic conditions at the time which may have resulted in a higher concentration of green turtles (Rivas-Zinno, 2012).
Incidental captures of juvenile green turtles have also been documented on important foraging grounds off Argentina, especially Samborombón Bay and El Rincón, primarily from gill nets used by the artisanal fisheries, but also from shrimp nets and other artisanal fishing gear (González Carman et al., 2011). Green turtles utilizing foraging grounds off Argentina have been demonstrated to contain primarily individuals from the Ascension Islands nesting beaches. However, individuals from Trindade Island, Suriname, and Aves Island nesting assemblages were also utilizing the Argentine foraging grounds (Prosdocimi et al., 2012). Therefore impacts to green turtles off Argentina affect a variety of nesting assemblages within the western and central South Atlantic.

Drift gill net fishing off Brazil, primarily for hammerhead sharks, has been shown to be a source of incidental capture and mortality of sea turtles. While green turtles were the least impacted of the three species observed (leatherback and loggerheads were the other two), it was estimated that a minimum of 134 green turtles were caught, and a minimum of 30 killed, per year in that one fishery (Fiedler et al., 2012). For all of the gill net studies indicating mortality levels from dead individuals found in the nets, the actual mortality level is likely higher. Post-release mortality was found to be somewhere between 7 and 29 percent in a North Carolina study using shallow-set gill nets and short soak times of 4 hours (Snoddy and Southwood Williard, 2010). The Brazilian hammerhead gill net fishery frequently has soak times up to 12 hours (Fiedler et al., 2012).

Throughout the Caribbean areas of the South Atlantic DPS, both South American and insular nations, coastal fisheries such as gill nets, fish and lobster pots, and trawls present a substantial threat of incidental take of sea turtles, including green turtles (Dow et al., 2007).

In the eastern South Atlantic, sea turtle bycatch in fisheries has been documented from Gabon to South Africa (Fretey, 2001). Coastal fisheries implicated in bycatch of sea turtles include gill nets, beach seines, and trawlers (Bal et al., 2007). Fishing in the Gulf of Guinea, an important green turtle foraging ground, is known to take green turtles. In one study, 12 of 200 females tagged at the nesting beach were reported captured by fisheries in the Gulf of Guinea, with a mortality rate of 75 percent, within only a three-year period (Tomás et al., 2010). Given the likelihood of under reporting of take, as well as tag loss, the actual capture rate may be even more severe.

Industrial trawling off Guinea-Bissau is prosecuted by a variety of countries and the national government does not have any requirements for turtle excluder device use in their waters. There is also extensive illegal fishing occurring (Catry et al., 2009). Other gear such as gill nets also take sea turtles in the area (Catry et al., 2009). While the Bolama-Bijagós Biosphere Reserve covers the entire archipelago and provides some protection through the management of the reserve and the survey work patrolling the areas, limited enforcement and resource shortages somewhat limit the effectiveness of the reserve.

In Ghana, fishing is one of the primary trades of people living on the coast. However, fish stocks have been reduced through overfishing and environmental degradation and many fishermen that incidentally catch sea turtles will keep and kill the turtle to feed their families (Tanner, 2013).
Off another important West African nesting area on the Ivory Coast incidental catch of sea turtles, including juvenile greens, was said to be common. In 2001, a four-month period of observation at one fishing market revealed 18 slaughtered turtles, including three green turtles (Peñate et al., 2007). At that time sea turtle protection legislation was not respected and enforcement was almost non-existent. Since then, a push has been made to generate alternative sources of income for the local populations and to employ ex-poachers to patrol the beaches (Peñate et al., 2007).

Fishing effort off the western African coast has been increasing (see Palomares and Pauly, 2004). Impacts continue to increase in the Guinea Current LME, but, in contrast, the impacts are reported to be decreasing in the Canary Current LME (http://www.lme.noaa.gov). Throughout the region, fish stocks are depleted and management authorities are striving to reduce the fishing pressure.

Pollution and Oil Exploration/Extraction

Various studies have shown high prevalence of marine debris ingestion by green turtles in the western South Atlantic, in some cases occurring in 100 percent of the individuals examined (Bugoni et al., 2001; Tourinho et al., 2010; Guebert-Bartholo et al., 2011; Murman, 2011). While the sublethal effects of such ingestion are hard to quantify, mortality directly attributable to ingestion of marine debris was estimated to be about 13 percent by Bugoni et al. (2001) and 9 percent by Tourinho et al. (2010), and as high as 56 percent by Murman (2011). Similar impacts from marine debris can be expected in the Caribbean and Atlantic Africa as coastal populations continue to grow and plastic use increases.

The direct impacts of pollution on green turtles’ health and survivorship are often difficult to quantify, or often even describe. However, polluted waters have the potential to cause various problems for sea turtles, including increasing the likelihood of contracting diseases such as FP as detailed earlier. In the coastal waters off Suriname, especially the northwest, fertilizer and pesticide run-off from agricultural activities higher in the watershed can be extensive (Reichart and Fretey, 1993).

Oil reserve exploration and extraction activities also may pose a threat for sea turtles in the South Atlantic. Seismic surveys in Brazil and Angola have recorded sea turtle occurrences near the seismic work (de Gurjao et al., 2005; Weir, 2007). While no sea turtle takes were directly observed on these surveys, increased equipment and presence in the water that is associated with these activities also increases the likelihood of sea turtle interactions (Weir, 2007). Oil exploration and extraction within the Gulf of Guinea rapidly increased since the discovery of oil reserves in the past two decades (Formia et al., 2003), with the associated activities and potential for oil spills and other pollution creating a threat to the important foraging areas and nesting beaches for green turtles in the area.

Climate Change

As in other areas of the world, climate change and sea level rise have the potential to impact green turtles in the South Atlantic. This includes beach erosion and loss from rising sea levels,
skewed hatchling sex ratios from rising beach incubation temperatures, and abrupt disruption of ocean currents used for natural dispersal during the complex life cycle. In very low-lying islands such as Aves, rising sea levels and increased storms could potentially eliminate it's functionality as a nesting beach. Evidence from green turtles nesting in Ascension Island indicates that adaptive differentiation in nesting behavior can occur even in single-island populations encountering different thermal conditions at different beaches. Adaptive nesting behavior may ameliorate somewhat the impact of increased temperatures resulting from climate change (Weber et al., 2012). This does not, however, address the issue of sea level rise impacts. Climate change impacts could have profound long term impacts on nesting populations in the South Atlantic, but it is not possible to quantify the potential impacts at this point in time.

Natural Disasters

Natural environmental events may affect green turtles in the South Atlantic. Parts of the South Atlantic DPS region in the Caribbean are susceptible to hurricane impacts. In general, however, severe storm events are episodic and, although they may affect green turtle hatchling production, the results are generally localized and they rarely result in whole-scale losses over multiple nesting seasons. This is generally due to higher wind speeds aloft, preventing the storms from gaining height and therefore strength. However, a rare hurricane hit Brazil in March 2004, which is atypical in the western or eastern South Atlantic (McTaggart-Cowan et al., 2006).

7.2.6. Summary of Existing Conservation Efforts

The main threats to South Atlantic green turtles include fishery bycatch, marine debris and pollution, habitat destruction impacting eggs and hatchlings at nesting beaches, and nest and hatchling predation. Most South Atlantic countries, including those in South America, the Caribbean, and Africa have developed national legislation and have various projects sponsored by governments, local communities, academic institutions, and non-governmental organizations to protect sea turtles, and nesting and foraging habitats to varying degrees (Formia et al., 2003; Dow et al., 2007). The consistency and effectiveness of such programs likely vary greatly across countries and over time based on resource availability and political stability. In addition, some countries have site specific legislation or conservation designation for turtle habitat protection.

Conservation through education is a widely-used and valuable tool throughout nations within the South Atlantic DPS and around the world. Such education initiatives can be highly successful. In Akassa, Nigeria, dedicated, intensive conservation education program by the Akassa Community Development Project resulted in sea turtles being recognized locally as an essential part of the area’s natural heritage. This has resulted in the majority of the nests in Akassa being protected, and when live stranded turtles are found they are released (Formia et al., 2003). However, in areas where the utilization of sea turtles is deeply ingrained in the local culture, such as the La Guajira region of Colombia (Patino-Martinez et al., 2012) changing people's attitudes about the use of sea turtles can be a long, slow process.

In the Caribbean South Atlantic green turtle conservation on the nesting beach varies widely among the 22 nations and territories. However, programs at what are by far the three largest nesting sites; Aves Island, French Guiana, and Suriname, with over 500 crawls per year (Dow et
provide protection to a significant proportion of nesting in the area. Aves Island has been established as a protected wildlife refuge by Venezuela since 1972. Vera and Buitrago (2012) determined that although nesting is slowly increasing on Aves, at the current rate of increase it would take 150 years to reach the historical levels of abundance. In French Guiana, the destruction or poaching of nests, eggs, or sea turtles was strictly prohibited by 1991 regulations passed by France under the Protection of Nature Act of 1976. In Suriname, the primary green turtle nesting beaches are protected within nature reserves, Matapica Beach in the north is within the Matapica Nature Reserve, and Babunsanti is in the Galibi Nature Reserve. In Suriname sea turtles and their eggs are fully protected by law except for a limited allowance of traditional harvest (Dow et al., 2007), though poaching remained a problem after the protective measures were put in place (Reichart and Fretey, 1993). While the conservation efforts at the largest nesting sites in the Caribbean are substantial there are monitoring and enforcement limitations in those areas (Dow et al., 2007). Additionally, some smaller, but still important nesting sites in the region do not have the equivalent conservation efforts.

In South America, outside of the Caribbean, Brazil is the only nation with substantial green turtle nesting. In Brazil the primary nesting areas are monitored by Projeto TAMAR, the national sea turtle conservation program in Brazil. Since 1980, Projeto TAMAR has worked to establish legal protection for nesting beaches (Marcovaldi and dei Marcovaldi, 1999). As such many of the detrimental human activities described previously are restricted by various state and federal laws (Marcovaldi and dei Marcovaldi, 1999; Marcovaldi et al., 2002; 2005). Nevertheless, tourism development in coastal areas in Brazil is high, and Projeto TAMAR works toward raising awareness of turtles and their conservation needs through educational and informational activities at their Visitor Centers that are dispersed throughout the nesting areas (Marcovaldi et al., 2005, Marcovaldi 2011). Since 1990 Tamar has worked along green turtle foraging areas as Almofala and Ubatuba (Marcovaldi et al. 2002). In 2001 the Brazilian Plan for Reduction of Incidental Sea Turtle Capture in Fisheries was created to address incidental capture of the five species in the country (Marcovaldi et al. 2002, Marcovaldi et al. 2006). The National Action Plan for the Conservation of Sea Turtles was published in 2010 by the Brazilian environmental government ICMBio (http://www.icmbio.gov.br/portal/images/stories/docs-plano-de-acao/pan-tartarugas/livro_tartarugas.pdf).

Green turtle nesting occurs on many beaches along the western coast of Africa, and there have been, and continue to be, sea turtle projects in many of the nations in the area ranging from research to public awareness to government conservation efforts (see Formia et al., 2003 for a regional synopsis). The largest nesting assemblages occur on Poilão, Bijagos Archipelago, Guinea Bissau, and on Bioko Island, Equatorial Guinea. While conservation efforts on the beaches have been established, issues with enforcement capabilities and resources make consistent protection problematic (Formia et al., 2003; Catry et al., 2009; Tomás et al., 2010).

Green turtle conservation efforts on Ascension Island have involved extensive monitoring, outreach, and research. The group Turtles in the UK Overseas Territories aims to progress the conservation, research and management of marine turtle populations and their habitats, and has worked extensively on Ascension Island (http://www.seaturtle.org/mtrg/projects/tukot/ascension.shtml). Additionally, there are legal prohibitions protecting sea turtles on Ascension.
Green turtles of various sizes and life stages occur throughout the South Atlantic. Within national waters of specific countries, various laws and actions have been instituted to mitigate threats to green turtles and other species of sea turtles; less protection is afforded in the high seas of the South Atlantic. Overall, the principal in-water threat to green turtles in the South Atlantic is incidental capture in fisheries. Within the Caribbean portion of the South Atlantic DPS region, conservation various conservation measures ranging from protected areas to education are utilized to protect green turtles (Dow et al., 2007).

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 incidental capture 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). The Karumbe conservation project in Uruguay has been working on assessing in-water threats to marine turtles for several years (see http://www.seaturtle.org/promacoda), with the objective of developing mitigation plans in the future. In Argentina, various conservation organizations are working toward assessing bycatch of green turtles and other sea turtle species in fisheries, with the objective of developing mitigation plans for this threat (http://www.pricetma.com.ar).

Coastal fisheries along western Africa is a major source of sea turtle mortality and several authors have highlighted the need to develop regional mitigation plans to reduce bycatch of green turtles and other sea turtle species in coastal waters (Formia et al., 2003; Weir et al., 2007; Peterson et al., 2009). Conservation strategies including marine protected areas, education, and community outreach have been implemented to help in reducing impacts to green turtles in their coastal habitats from bycatch and intentional take (Formia et al., 2003).

Overall, conservation efforts for green turtles in the South Atlantic DPS are inconsistent. While there are numerous varied conservation efforts, especially on the primary nesting beaches, many issues remain due to limited enforcement of existing laws and marine protected areas as well as extensive fishery bycatch, especially in coastal waters. The effectiveness and consistency of conservation measures will need to be increased substantially to prevent the further decline, and allow the recovery of, this DPS in the future.

15.1.4.6. National Legislation and Protection

In addition to the international mechanisms, most South Atlantic countries have developed legislation to protect sea turtles and/or nesting habitats. They are summarized below. However, the overall effectiveness of this country specific legislation is unknown at this time.

Note: A more complete account of protective national legislation in Atlantic African nations can be seen in (Fretey, 2001).
**Angola**

All sea turtles are granted full protection under the 1972 amendment to the 1957 Hunting Regulations.

**Argentina**

Sea turtles are provided protection generally under Law 22.421, the Wildlife Law for National Territories. Various other decrees and laws, both national and regional regulate fisheries and other activities to limit sea turtle impacts. In 2007 Resolution SA and DS 513/07 was enacted that explicitly prohibits hunting, capture, transporting between provinces, trade at the federal jurisdictional level and exporting live specimens, products and subproducts of wildlife, which includes sea turtles in its Annex I.

**Ascension Island**

Green turtles in Ascension Island are locally protected by the Wild Life Protection Ordinance of 1944 and the Wild Life Protection Regulations of 1967. The 1944 ordinance specifies a schedule of species to be protected, and includes sea turtles and turtle eggs. The 1967 ordinance more broadly defines the Governor’s power to prohibit the killing, capturing, or taking of any wildlife on the island. Furthermore, the Endangered Species Control Ordinance of 1976 controls certain imports and exports to and from Ascension Island.

**Benin**

Although the designation of “giant sea tortoises” is on the list of protected species there is no reference to genus or species, and the general category of “chelonians” is included in the list of small game, thus making the legal value of sea turtle protection questionable in Benin (Fretey, 2001).

**Brazil**

The Brazilian Red List classifies green turtles as threatened/vulnerable (Almeida et al. 2011b). The Law on Environmental Crimes No. 9605 makes the harvest or consumption of sea turtles illegal. Brazil also has various other laws establishing fishing gear restrictions, lighting requirements near nesting beaches, traffic restrictions on nesting beaches, and regulating seismic survey and other oil exploration activity during nesting season. In 1980 Projeto TAMAR, a collaboration between the government, an NGO, the private sector, and local communities was formed to protect sea turtles through research, education and outreach, community involvement, and enforcement of sea turtle protection regulations.

**British Virgin Islands**


Comment [A29]: Response: accept changes
The green turtle is listed as “endangered” under the First Schedule of the 1976 British Virgin Islands Endangered Animals and Plants Ordinance, which prohibits their importation and exportation.

Cameroon

Green turtles are protected under a variety of fishery and wildlife regulations.

Cape Verde

Decreto n°97/87 : September 5, 1987 (Law 06/94): Article 17: Prohibition of capturing sea turtles from the 1st of July until the end of February (Fretey, 2001).

Colombia

Green turtles in Colombia are protected by a number of regulations, both national and region-specific within the country. TED use is required in, and technical specifications are established under, Resoluciónes No. 108 (1992), 157 (1993), 148 (1994), 107 (1996), and 68 (1999). 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).

Congo

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 Conkaouati-Douli National Park. However, in areas without permanent monitoring almost all eggs and nesting individuals are collected and eaten (Bal et al., 2007).

Democratic Republic of the Congo

Green 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).

Equatorial Guinea

Since 1988 Equatorial Guinea legally protected all sea turtles under Law 8/1988 and Decree 183/87 on fishing (Tomás et al., 2010). However, despite that law egg harvest and active, organized nesting female harvest for local consumption and sale has occurred (Castroviejo et al., 1994).

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).

*Gabon*

Since 2011 Gabon has decreed protection for all sea turtle species (http://www.seaturtle.org/groups/gabon/home.html). There are five national parks in Gabon that protect sea turtle habitat.

*The Gambia*

Wild animals cited in the laws, including sea turtles, are protected under the Wildlife Conservation Act (1977) and the Wildlife Conservation Regulations (1977).

*Ghana*

The Wildlife Regulations Act of 1974 protects all sea turtle species in Ghana from poaching and egg harvest. In general the populace is reluctant to break the law for fear of stiff penalties and there is little commercial use of sea turtles, but poverty is prevalent and it is not unusual for individuals to capture and kill nesting sea turtles encountered on the beach, to be used for personal consumption (Tanner, 2013).

*Guinea-Bissau*

On the Bijagós Archipelago, one of the largest green turtle nesting sites in Atlantic Africa, all marine turtles are protected under a variety of national laws and regulations, but enforcement is limited.

*Guinea*

Green turtles are granted protection under a variety of wildlife protection acts and fisheries regulations.

*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.

*Ivory Coast*
Green turtles are granted protection under a number of national laws, including as part of Annex I which provides full protection to all Chelonidae species and prohibits the taking of eggs turtles.

**Liberia**

Green turtles are listed in Annex II, classified as fully protected, in a 1976 act establishing the Forestry Development Authority.

**Namibia**

While not specifically listed in the Nature Conservation Ordinance of 1975, all sea turtle species are fully protected in proclaimed conservation areas (which encompass 95% of the coast). Additionally, sea turtles are protected under the Sea Fisheries Regulations.

**Nigeria**

Decree No. 11 (Endangered Species, Control of International Trade and Traffic) prohibits the hunting, capture, or trade of animal species threatened with extinction. Green turtles are the only species officially protected (Fretéy, 2001). Nigeria does require TED use in their shrimp trawl fishery and participates in the U.S. section 609 TED certification program.

**St. Helena**

Has no legislation specific to sea turtles but gain protection through general legislation including the Protection of Animals Ordinance and the Wildlife Protection Ordinance.

**Sao Tome and Principe**

Green turtles are protected under a variety of fishery and wildlife regulations.

**Senegal**

The Code des Peches Maritimes (1976) prohibits the taking, possession, and sale of all species of sea turtles. Additional later regulations specifically prohibited the taking of young sea turtles and collecting eggs.

**Sierra-Leone**

Green turtles are protected under a variety of wildlife and fishery laws, including The Wildlife Conservation Act (1997).

**South Africa**

All sea turtle species are protected and may not be killed, molested, or traded per Ordinance 15, Section 101, 1974.
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 1992 the government decreed that TED use was mandatory on all shrimp trawl vessels.

Togo

Green turtles are protected under wildlife and fishery laws in Togo.

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 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). Subsequently, 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.’

Turks and Caicos Islands

The Fisheries Protection Ordinance, Cap. 104 (1995) is the main legislation which provides the legal basis and regulations for managing the fishery resources of the Turks and Caicos Islands. It imposes a size limit for turtles (Fleming, 2001).

U.S. Virgin Islands

In addition to the ESA, the U.S. Virgin Islands Code, Chapter 9A, Title 12, Section 318 protects sea turtles, nests and eggs (1972). 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).

Uruguay

Sea turtles in Uruguay are protected by presidential decree (144/998, June 1998) which prohibits the capture, retention, transport, commerce, transformation or processing of sea turtles. The import or export of sea turtle products is regulated by law number 14.205 which follows CITES on trade of protected species.
Venezuela

Aves Island was designated as a marine reserve in 1972, providing full protection to sea turtles and their nests on and around the island. 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 closing hunting on all species in danger of extinction among the fundamental sea turtle protection measures.

15.1.4.7. International Instruments

At least 20 regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the South Atlantic. The international instruments listed below apply to sea turtles found in the South Atlantic and are described in Appendix 5.

- African Convention on the Conservation of Nature and Natural Resources (Algiers Convention)
- Convention on Biological Diversity
- Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention)
- Convention on the Conservation of European Wildlife and Natural Habitats
- Convention on the Conservation of Migratory Species of Wild Animals
- 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)
- Convention on International Trade in Endangered Species of Wild Fauna and Flora
- Food and Agriculture Organization Technical Consultation on Sea Turtle-Fishery Interactions
- Inter-American Convention for the Protection and Conservation of Sea Turtles
- International Convention for the Prevention of Pollution from Ships (MARPOL)
- International Union for Conservation of Nature
- Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa (Abidjan Memorandum)
- Protocol Concerning Specially Protected Areas and Wildlife
- Ramsar Convention on Wetlands
- South-East Atlantic Fisheries Organization
- United States Magnuson-Stevens Fishery Conservation and Management Act
- United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing
7.3.15.2. Assessment of Significant Portion of its Range (SPR)

The SRT reviewed the information on threats and extinction risk within each DPS to determine whether there were portions of the range of the DPS that might be considered "significant portions of the range" in accordance with the definition of endangered and threatened species and the draft policy that interprets that term (see Section 3.4).

This DPS has nesting in three geographic regions: (1) West Africa, (2) Central South Atlantic (Ascension Island), and (3) South America. It was acknowledged that there was a high level of uncertainty about the severity of threats, annual nesting abundance, and effectiveness of conservation efforts and enforcement of regulations along West Africa. Threats in this West Africa region are likely greater than threats in the other regions within the South Atlantic DPS. However, the SRT concluded that, even if threats were so great that the West African populations were lost (not necessarily likely, but the test for an SPR), the DPS would not be at a substantially higher risk of extinction. As such, the SRT concluded that the need to consider a significant portion of the range does not apply to this DPS.

15.3. Assessment of Extinction Risk

For the SRT’s assessment of extinction risk for green turtles in the South Atlantic DPS, there were two separate sets of ranking exercises: One focusing on the importance that each SRT member placed on each of the six different elements for this region (Table 7.3), and a second which reflects the SRT members’ expert opinion about the probability that green turtles would fall into any one of various extinction probability ranges (Table 7.4). See Section 3.3. for details on the six elements and the voting process.

**Table 7.3.** Summary of ranks that reflect the importance placed by each SRT member on the critical assessment elements considered for the South Atlantic DPS. For Elements 1–4, higher ranks indicate higher risk factors.

<table>
<thead>
<tr>
<th>Critical Assessment Elements</th>
<th>Element 1</th>
<th>Element 2</th>
<th>Element 3</th>
<th>Element 4</th>
<th>Element 5</th>
<th>Element 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abundance (1 to 5)</td>
<td>MEAN RANK</td>
<td>1.58</td>
<td>1.92</td>
<td>1.33</td>
<td>1.67</td>
<td>-0.83</td>
</tr>
<tr>
<td></td>
<td>SEM</td>
<td>0.19</td>
<td>0.15</td>
<td>0.14</td>
<td>0.14</td>
<td>0.11</td>
</tr>
<tr>
<td></td>
<td>RANGE</td>
<td>1–3</td>
<td>1–3</td>
<td>1–2</td>
<td>1–2</td>
<td>−1–0</td>
</tr>
</tbody>
</table>

With respect to the importance rankings for the six elements, the first four elements using the 1-5 ranking system (higher rank equals higher risk factor), no one element stood out. The range of mean ranks was 1.33 to 1.92 for the four elements, indicating very low to low risk for each of those individual elements contributing significantly to risk of extinction for the DPS. Spatial structure (i.e., widespread overall nesting distribution) featured relatively low (1.33) in the risk
threshold voting, likely resulting from the geographically widespread nature of the DPS, along with substantial nesting beaches occurring across the DPS as opposed to being limited to one area of the DPS. The highest risk (1.92) was for trends/productivity. This likely reflects the fact that while some of the largest nesting beaches such as Ascension Island, Aves Island, and Galibi appear to be increasing, others such as Trindade Island, Atol das Rocas, and Poilão and the rest of Guinea-Bissau seem to be stable or do not have sufficient data to make a determination, and Bioko appears to be in decline.

SRT members also generally thought that future threats not yet reflected in nester abundance or not yet experienced by the population weighed slightly heavier in their risk assessment voting than any conservation efforts are not yet reflected in nester abundance. With respect to the diversity of opinions among the SRT members when considering the six Critical Elements, a relatively large range in rankings (i.e., voter opinion) was noted for the abundance and trends/productivity elements (w/ ranks from 1 to 3).

Table 7.4. Summary of Green Turtle SRT member expert opinion about the probability that the South Atlantic DPS will reach a critical risk threshold within 100 years. Each SRT member assigned 100 points across the rank categories. The continuum in Row 1 has categories with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1-5%</th>
<th>6-10%</th>
<th>11-20%</th>
<th>21-50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN ASSIGNED POINTS</td>
<td>69.00</td>
<td>16.50</td>
<td>9.92</td>
<td>4.17</td>
<td>0.42</td>
<td>0.00</td>
</tr>
<tr>
<td>SEM</td>
<td>9.05</td>
<td>3.43</td>
<td>4.18</td>
<td>2.29</td>
<td>0.42</td>
<td>0.00</td>
</tr>
<tr>
<td>Min</td>
<td>10</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Max</td>
<td>98</td>
<td>40</td>
<td>45</td>
<td>20</td>
<td>5</td>
<td>0</td>
</tr>
</tbody>
</table>

Of the categories describing the probability that the South Atlantic DPS will reach a critical risk threshold within 100 years, SRT members voted overwhelmingly for the two lowest probability designations, with 69 percent of the votes in the '<1%' range and 16.5 percent of the votes in the '1–5%' range. No votes were cast for the highest range (>50%) and only 0.42 percent of the votes were cast in the '21–50%' risk range.

In the vote justifications, a widespread geographical range, along with high abundance nesting sites spread across that range, were typically cited as influential factors. The prevalence of both insular and mainland nesting sites was also cited by some members. Concerns that were shared by multiple members included the uncertainty in trends at some of the more important nesting beaches due to data deficiencies, the fact that some of the larger rookeries are not showing increases, and the disparate but continuing threat levels for significant portions of the population.

15.4. Synthesis and Integration
During the analysis of the South Atlantic DPS’s status an integrated approach was taken by the SRT to consider the many critical elements described earlier. Nesting abundance for this DPS is relatively high, with large rookeries spread out geographically and a large number of nesting sites being used by anywhere from a few females to tens of thousands of females. Population trends within those rookeries were inconsistent, and in many cases the data was limited and a trend could not be determined, even for major rookeries. While not a critical concern for the SRT, it was still one of the most cited worries, due to some important rookeries having either limited data, creating uncertainty, or showing stable or declining trends. While some of the largest nesting beaches such as Ascension Island, Aves Island, and Galibi appear to be increasing, others such as Trindade, Atol das Rocas, and Poilão and the rest of Guinea-Bissau seem to be stable or do not have sufficient data to make a determination, and Bioko appears to be in decline.

Spatial structure carried the lowest mean rank from the SRT votes, indicating the lowest level of concern with that element increasing extinction risk. The diversity/resilience of the DPS is bolstered by the widespread nature of the rookeries, ranging in nesting abundance from small to very large. A potential concern is the domination of the DPS by insular nesting sites, which has the potential to reduce the resilience of the DPS in the face of sea level rise and increasing tropical storm activity.

While abundance, population trends, spatial structure, and diversity/resilience were considered by the SRT to have a low likelihood of contributing to the extinction of the DPS in the next 100 years, many concerns remain in terms of outside threats. Habitat destruction/degradation both at nesting beaches and important foraging grounds is a continuing concern, though inconsistent across the DPS. Overutilization of green turtles within the South Atlantic was likely a primary factor in past declines. While reduced from those levels due to increased legal protections, it is still thought to be a low to moderate threat to the DPS. Within the DPS the threat can vary widely, from being very low for the insular nesting sites of Brazil, to being fairly extensive in some areas of western Africa. Disease and predation are continuing threats but not considered a primary threat to the DPS. Despite increasing legal protections for sea turtles within the DPS the inadequacy of existing regulatory mechanisms is a noted issue. While many international and national laws purporting to protect sea turtles exist, limitations in resources and political will creates a situation of inconsistent or sometimes nonexistent practical measures to enforce those laws. Fishery bycatch also continues to be a major concern throughout the DPS, near nesting beaches and foraging areas as well as on the high seas. Increasing awareness and conservation efforts by governments, local communities, non-governmental organizations, and industries has helped to reduce threats, but remains inconsistent and often resource limited.

While overall the SRT determined the likelihood of reaching a critical risk threshold within 100 years was relatively low (69 percent of votes cast for the ‘<1%’ likelihood category), there was still a sizeable percent of votes cast for the categories from 1-20 percent likelihood: 16.5 percent of the votes for a ‘1-5%’ likelihood; 9.92 percent for the ‘6-10%’ likelihood; 4.17 percent for the ‘11-20%’ likelihood; and even a very small number (0.42 percent) cast for the ‘21-50%’ likelihood category. These results reflect the view that while the DPS shows strength in many of the critical assessment elements, there are still concerns about the fairly large uncertainty about
trends and threat impacts to many important nesting sites, as well as the ongoing threats known to affect the DPS.
8. SOUTHWEST INDIAN DPS (DPS #4)

8.1. DPS Range and Nesting Distribution

The Southwest Indian DPS encompasses Madagascar as well as a number of island nations in the western Indian Ocean. Its western boundary is marked by the shores of continental Africa from just north of the Kenya-Somalia border (0°) south to the Cape of Good Hope (South Africa), extending from there to 19°E, 40°S; its northern boundary extends from just north of the Kenya-Somalia border along the equator eastward to the 84°E; its eastern boundary runs along the 84°E parallel from the equator to 40°S latitude; and its southern boundary extends from 40°S, 19°E to 40°S, 84°E. Nesting occurs along the east coast of Africa as far south as 25°S, the north, west, and south coasts of Madagascar, and scattered offshore islands in the southwest Indian Ocean (Figure 8.1). Nesting hotspots are the French Eparses Islands (Europa, Glorieuses, and Tromelin), Mayotte and the outer Seychelles islands (Aldabra group including Aldabra, Assumption, Cosmoledo, and Astove); Farquhar; and Amirantes Group; Bourjea, 2012a; Mortimer 1984; Table 8.1).

![Figure 8.1. Nesting distribution of green turtles in the Southwest Indian DPS. Size of circles indicates nesting abundance category. Locations marked with '×' indicate nesting sites lacking abundance information.](image-url)
Foraging occurs along the east coast of Africa, around Madagascar where numerous sea grass beds are found, and on shallow banks and shoals throughout the region, including those associated with virtually every island in Seychelles (Mortimer 1984, Mortimer et al. 1996). Small and immature animals are also concentrated around Bazaruto and Inhassoro and some found in Maputo Bay (Bourjea, 2012). Along the coast of Kenya, an aerial survey in 1994 indicated that sea turtles are widely distributed within the 20 m isobath mainly within sea grass beds and coral reefs (Frazier, 1975; Wamukoya et al., 1996; Okemwa et al., 2004). The eastern seaboard of South Africa serves as a feeding and developmental area for green turtles (Bourjea, 2012).

8.2. Critical Assessment Elements

In the evaluation of extinction risk for green turtles in the Southwest Indian DPS, the SRT considered six distinct elements, each of which are discussed below: (1) Nesting Abundance, (2) Population Trends, (3) Spatial Structure, (4) Diversity / Resilience, (5) Five-Factor Threat Analyses, and (6) Existing Conservation Efforts.

8.2.1. Nesting Abundance

The vast majority of nesting occurs on isolated islands but also reported along much of the Madagascar and East African coasts as far south as 25°S (Bourjea, 2012). For the DPS, there are 17 rookeries with some measure of abundance, where four of which consisted of more than 10,000 nesters (~30 percent of the total adult females; Table 8.2).

For the four Comoros Islands (Grande Comore, Mohéli, Anjouan, and Mayotte), monitoring of green turtle nesting is conducted at Mohéli and Mayotte islands. Mohéli Island (12°15’S, 43°45’E) is located in the north of the Mozambique Channel. Assisted by its Marine Protected Areas status, the beaches of Mohéli at Itsamia are among the most important nesting sites for this species in the southwest Indian Ocean (Frazier, 1985; Bourjea, 2012). Mayotte Island (12°50’S, 45°8’E) is located in the north of the Mozambique Channel and it is the eastern most island of the Comoros Archipelago. Nesting on the island occurs throughout the year, though a nesting peak occurs in June. There are 170 beaches that are suitable for turtle nesting. Green turtle nesting is regularly monitored on Saziley National Park in Mayotte. The area has six discrete beaches with 2239 m of sandy beach suitable for nesting, where the largest beach is Grande Saziley. Grand Saziley has been monitored nightly since January 1998 (Bourjea et al., 2007b).

The Eparses Islands are French islands scattered in the southwest Indian Ocean around the coast of Madagascar. Three of the islands are important nesting sites for green turtles: Tromelin, Les Glorieuses, and Europa (Le Gall et al., 1986; Lauret-Stepler et al., 2007). Tromelin Island (15°33’S, 54°31’E) is outside of the Mozambique Channel. It lies 560 km north of Reunion Island and 470 km east of Madagascar. Most of the coast is covered with boulders, but there is a sandy beach (approximately 1,600 m in length) suitable for turtle nesting in the northwestern part and has been monitored daily since March 1986 (Lauret-Stepler et al., 2007). Grande Glorieuse (11°33’ S, 47°17’ E) is the largest island in the Les Glorieuses archipelago in the northern Mozambique Channel, 220 km from Madagascar. The island is divided into two sampling zones: a 1,500 m stretch of beach between the military base and the landing stage
(approximately 16 percent of suitable nesting habitat on the island), and the rest of the island. The former zone has been sampled daily since January 1987, whereas the rest of the island has been monitored since January 2001 (Lauret-Stepler et al., 2007). Europa (22°21’S, 40°21’E) is the largest of the Eparses Islands and lies in the southern Mozambique Channel, 330 km from Madagascar. Daily nesting surveys have been conducted since June 1983 on the 1600 m stretch of beach, representing approximately 26 percent of the sandy beaches suitable for nesting turtles (Lauret-Stepler et al., 2007). Nesting also occurs on La Reunion, an island east of Madagascar.

The Republic of Seychelles is a 115-island country spanning an archipelago in the Indian Ocean east of mainland Africa and northeast of Madagascar. Aldabra Atoll (9°24’S, 46°20’E), part of the Outer Islands of the Seychelles, is located at the north end of the Mozambique Channel. It is a slightly elevated coral reef on the summit of a volcanic peak rising from a depth of 4000 m (Mortimer et al., 2011). It is a UNESCO World Heritage Site managed by the Seychelles Islands Foundation and has four main islands with a total outer perimeter of 83 km, of which 5.2 km is nesting habitat consisting of coralline sand (52 beaches). The beaches occur within six beach groups; two on the west coast, one in the north, and three on the south coast.

Based on a study from 1997 to 2000, 684 nests were recorded, of which green turtles made up 94 percent of the nesting activity, with the remainder comprising hawksbill and olive ridley nests (Okemwa, 2003 cited in Bourjea, 2012a). Along the coast of Madagascar, green turtles nest on beaches of the north, south and west Malagasy coast. Few nesting records are known from the east side of Madagascar. Sea turtle nesting is common on all the outer islands of St. Brandon, Agalega, and Chagos. However, few nests are found on Mauritius and Rodrigues, presumably depleted after years of development and disturbance (Bourjea, 2012). Green turtles nest predominantly north of the tropic of Capricorn, from Quewene Peninsula to the Quirimbas Archipelago, with the main concentrations of nesting in the Primeriras and Segundas Islands and Quirimbas Archipelago (Vamizi and Rongui Islands). Madagascar is also known to be an important feeding area for sea turtles (Bourjea, 2012).

In Tanzania, although exact nesting abundance is unknown, important nesting sites for green turtles in Zanzibar are Misali (west), Vumawimbi, and Kiuyu in Pemba; and Matemwe and Mnemba Islands in Unguja. Other key turtle nesting sites of relative importance are Mafia (high), Temeke (medium), Mtwara (low) and Pangani (medium). An average of 450 green turtle nests are recorded per year. However, these numbers only represent data for part of the Tanzania’s mainland coastline.
Table 8.1. Summary of green turtle nesting activity in the Southwest Indian DPS. Data are organized by country, nesting site, monitoring period, and estimated total nester abundance \[ \text{(Total Counted Females/Years of Monitoring) x Remigration Interval} \], and represents only those sites for which there were sufficient data to estimate number of females. Many nesting sites in the Southwest Indian DPS are data deficient and estimates could not be made for those beaches. The total female abundance was computed by multiplying the annual abundance by the assumed mean remigration interval (3 yrs). For a list of references for these data, see Appendix 2.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Republic of Seychelles</td>
<td>Aldabra Atoll</td>
<td>1981–2012*</td>
<td>16,000</td>
</tr>
<tr>
<td>Republic of Seychelles</td>
<td>Inner Islands</td>
<td>1981–2012*</td>
<td>~50 females nesting annually</td>
</tr>
<tr>
<td>Islamic Republic of Comoros</td>
<td>Mohéli</td>
<td>2000–2012*</td>
<td>15,000</td>
</tr>
<tr>
<td>France (Indian Ocean)</td>
<td>Mayotte</td>
<td>1998–2012*</td>
<td>12,000</td>
</tr>
<tr>
<td>France (Indian Ocean)</td>
<td>Tromelin</td>
<td>1987–2012*</td>
<td>4,500</td>
</tr>
<tr>
<td>France (Indian Ocean)</td>
<td>Europa</td>
<td>1986–2012*</td>
<td>25,500</td>
</tr>
<tr>
<td>France (Indian Ocean)</td>
<td>Glorieuses</td>
<td>1987–2012*</td>
<td>6,000</td>
</tr>
<tr>
<td>Madagascar</td>
<td>Nosy Iranja Kely</td>
<td>2003*</td>
<td>153</td>
</tr>
<tr>
<td>Kenya</td>
<td>Entire Coastline</td>
<td>2000*</td>
<td>1,500</td>
</tr>
<tr>
<td>Mozambique</td>
<td>Coastline and Islands</td>
<td>2004–2012*</td>
<td>150</td>
</tr>
<tr>
<td>Tanzania (including Zanzibar)</td>
<td>Zanzibar: Pemba, Unguja, Mnemba, Misali Islands</td>
<td>Sporadic since early 1990s</td>
<td>1,500</td>
</tr>
<tr>
<td>British Indian Ocean Territory (Mauritius)</td>
<td>Chagos Archipelago</td>
<td>1996, 1999, 2006*</td>
<td>1,800</td>
</tr>
</tbody>
</table>

* Monitoring at these sites is ongoing.
Table 8.2. Green turtle nester abundance distribution among nesting sites in the Southwest Indian DPS.

<table>
<thead>
<tr>
<th>NESTER ABUNDANCE</th>
<th># NESTING SITES DPS 4</th>
</tr>
</thead>
<tbody>
<tr>
<td>n/a</td>
<td>23</td>
</tr>
<tr>
<td>1–10</td>
<td>2</td>
</tr>
<tr>
<td>11–50</td>
<td>0</td>
</tr>
<tr>
<td>51–100</td>
<td>0</td>
</tr>
<tr>
<td>101–500</td>
<td>3</td>
</tr>
<tr>
<td>501–1000</td>
<td>7</td>
</tr>
<tr>
<td>1001–5000</td>
<td>1</td>
</tr>
<tr>
<td>5001–10000</td>
<td>1</td>
</tr>
<tr>
<td>&gt;10,000</td>
<td>4</td>
</tr>
<tr>
<td><strong>TOTAL NESTING SITES</strong></td>
<td><strong>17</strong></td>
</tr>
<tr>
<td><strong>TOTAL ABUNDANCE</strong></td>
<td><strong>84,199</strong></td>
</tr>
<tr>
<td><strong>PERCENTAGE AT LARGEST NESTING SITE</strong></td>
<td><strong>30% (Europa, Eparses Islands)</strong></td>
</tr>
</tbody>
</table>

8.2.2. Population Trends

Among the five species of sea turtles found in the area, the green turtle is the most abundant sea turtle species in this region and is known to nest on beaches of most countries. Green turtles in the Southwest Indian Ocean were exploited for many decades (Hughes, 1974; Frazier, 1980, 1982; Mortimer et al., 2011). The species, however, has successfully recovered at some nesting beaches in the recent years and trend data show growing trends, albeit largely at protected sites (Bourjea, 2012). For a list of references on trend data, see Appendix 3.

In Kenya, approximately 200–300 females nested each year from 1999 to 2004 (Okemwa and Wamukota, 2006); however, there are not sufficient data to determine the current population trend.

At the Comoros Islands, there were approximately 1,850 females per year in the early 1970s (Frazier, 1985), and about 5,000 females in 2000 (S. Ahamada, AID Environment-Comoros, pers. comm., 2001). However, because the more recent datum is unverified, this change in reported abundance should be viewed with caution. A newly monitored nesting site (Grande Saziley beach on Mayotte) has shown an average of over 1,500 nesting green turtles per year (Bourjea, 2012). Although the observed rate of change in the annual nesting females from 1998 to 2005 was stable, the time series was too short for providing a statistically robust estimate of population growth at the nesting beach.

At the Seychelles, nesting green turtles are currently increasing at protected sites, although the population remains depleted relative to historic levels (Mortimer et al., 2011). The annual number of nesting females at Aldabra and Assumption during the early 1900s was approximately 12,000 females based on information collected during the organized exploitation of the species for calipee production (Mortimer, 1985), and by the onset of protective measures in 1968 that number had dropped to an estimated less than 1,000 females (Mortimer, 1984). Since then,
however, the number of females nesting in the Seychelles has increased at protected sites. For example, at Aldabra, which is a nature reserve, the nesting activity increased from about 6038-8734 nests/yr during 1981 to 1984 to about 15,670 nests/yr from 2004 to 2008 (Mortimer et al., 2011).

In La Reunion Island, nesting started again in 2005 after a 25-years without nesting, 11 green turtle nests were recorded in a recent 3 year period (Ciccione and Bourjea, 2006; Bourjea, 2012).

At Aldabra Atoll of the Seychelles, the number of nests has increased 7 times in 40 years (Mortimer et al., 2011; Bourjea, 2012). At Mohéli of Comoros Islands, the annual population growth rate was 20 percent from 2000 to 2007 (Bourjea, 2012). At Mayotte, the annual growth rate was 0.9 percent from 1998 to 2006 (Bourjea et al., 2007b; Bourjea, 2012). At Tromelin, Eparses Islands, France, annual growth rate was -1.7% from 1986 to 2008 (Le Gall et al., 1986; Lauret-Stepler et al., 2007; Bourjea, 2012). At Europa, Eparses Islands, France, annual growth rate was 2 percent from 1986 to 2008 (Le Gall et al., 1986; Lauret-Stepler et al., 2007; Bourjea, 2012). At Glorieuses, French Eparses Islands, annual growth rate was 3.5 percent from 1987 to 2008 (Lauret-Stepler et al., 2007; Bourjea, 2012).

The green turtle shows overall large, stable or increasing nesting populations in the French Eparses Islands and Mayotte. At protected nesting sites with long-term monitoring indicated general increase in abundance, where five out of six monitoring sites have shown increase in nesting activities (Europa, Glorieuses, Mayotte, Mohéli, and Aldabra), whereas a declining trend has been reported for Tromelin (Bourjea, 2012). There are 3 nesting sites with greater than 10 years of recent monitoring data. These include Glorieuses, French Eparses Islands (19 yrs), Europa and Tromelin, Eparses Islands in France (20 yrs; Figure 8.3).

At Europa and Tromelin, the annual number of nesting females was reported at 4,000-5,000 in the early 1970s by Hughes (1970), and 9,000-18,000 in the late 1970s by Le Beau et al. (1979). Further, Le Gall et al., (1986) provided an estimate of 2,000-11,000 females per year for the mid-1970s to mid-1980s. More recent studies have used the number of tracks to compute long-term trends at these nesting sites (Lauret-Stepler et al., 2007). The nesting track records cannot be converted to nesting female abundance with a high level of certainty. Consequently, direct comparison to previous estimates of abundance is difficult.

There are three sites for which 10 or more years of recent data are available for annual nester abundance (the standards for representing trends in bar plot in this report; Figure 8.3). Of these, no sites met our standards for conducting a PVA (see Section 3.2 for more on data quantity and quality standards used for bar plots and PVAs). While true trends cannot be ascertained in many cases due to the lack of data, we discuss the indications of possible trends at some of the primary nesting sites.
8.2.3. Spatial structure

When examining spatial structure for the Southwest Indian DPS, the SRT examined three lines of evidence, including genetic data, flipper and satellite tagging, and demographic data.

Genetic sampling in the Southwest Indian DPS has been fairly extensive and the rookeries relatively well represented. However, sampling coverage of northern rookeries is lacking. Mitochondrial DNA studies indicate a moderate degree of spatial structuring within this DPS, with connectivity between proximate rookeries (see below). Overall, the Southwest Indian DPS appears to have at least two genetic stocks: (1) the South Mozambique Channel (SMC) consisting of Juan de Nova and Europa, and (2) the numerous rookeries in the North Mozambique Channel (NMC) consisting of Nosy Iranja, Mayotte, Mohéli, Glorieuses, Cosmoledo, Aldabra, Farquhar, also including Tromelin located east of Madagascar (Bourjea et

Comment [A16]: Jerome Bourjea and I are currently running a new analysis of green turtle samples from the Inner Islands and Amirantes Groups of Seychelles – both which are still exploited by people as they are near the human population centres. Initial findings suggest these rookeries may comprise a genetically distinct group – but it is still too early to state that here. Nevertheless, our initial findings suggest that the vastly depleted Inner Island/Amirantes green turtle rookeries may be very important.

Response: Good to know, but not sure we can use it here – next status review?
The authors suggested that the SMC stock could be further subdivided into two different genetic stocks, one in Europa and the other in Juan de Nova based on a significant haplotype frequency shift ($F_{ST} = 0.3030$). An Analysis of Molecular Variance further highlights the differentiation between the North and South Mozambique Channel and shows that 58.84 percent of the variance is found between the North and the South Mozambique rookeries.

Satellite telemetry data are available for green turtles that nest at some nesting beaches of this DPS. Green turtles nesting along the East African coast confine their migration to along the coast. This is in contrast to those nesting on islands (e.g., Comoros, Eparses, and Seychelles) which reach the East African or Malagasy coast via ‘migration corridors’ or nest on mid-oceanic sea grass beds. This behavior is believed to be mainly attributable to the fact those areas are characterized by a network of large seagrass beds (Bourjea, 2012).

Demographic information is available primarily on the nesting beaches of Seychelles, Moheli, Mayotte, and Tromelin, and La Reunion Island. The median CCL of nesters at Mayotte Island from 1998 to 2005 was 108 cm (Bourjea et al., 2007). The inter-nesting period ranges from 12-14 days at Mayotte (Frazier, 1985) and 12-13 days at Europa (Hirth, 1997) with an estimated remigration interval of at least three years (Mortimer et al., 2011). Reported clutch sizes vary for this DPS. In Seychelles, clutch sizes are 150-200 (Hirth, 1997). At Moheli, clutch sizes are 116 +/- 24 (Innocenzi et al., 2010). At Mayotte, the mean clutch size is 121.6 (Frazier, 1985). At Tromelin, mean clutch sizes range from 124.6 to 129 eggs (Hirth, 1997). At La Reunion Island, mean clutch size is 100 (SD=31.3, n=5, range=52-139; Ciccione and Bourjea, 2006). Incubation period at Reunion is >80 days in winter, 53 days in summer (Ciccione and Bourjea, 2006)) with a hatching success of >91 percent (Ciccione and Bourjea, 2006). Hatching success at Mohéli has been reported at 75.3% +/- 33.7 percent (Innocenzi et al., 2010).

8.2.4. Diversity / Resilience

The components considered under this critical element include the spatial range of nesting sites; diversity in nesting season, site structure, and orientation (e.g., high vs. low beach face, insular vs. continental nesting sites); and the genetic diversity within the DPS. These are important considerations for assessing the potential impact of catastrophic events such as storms, sea level rise, and disease.

The overall nesting range for the Southwest Indian DPS occurs throughout this DPS on islands, atolls, and on the main continent of Africa in Kenya. The nesting substrate can be variable as some of the nesting beaches are volcanic islands and the atolls made from coralline sand. Nesting occurs throughout the year with peaks that vary between rookeries (Dalleau et al. 2012; Mortimer 2012). The fact that turtles nest on both insular and continental sites suggests a high degree of nesting diversity.

The genetic structure of this DPS is characterized by high diversity and a mix of unique and rare haplotypes, as well as common and widespread haplotypes. These common and widespread haplotypes (CM-A8, CmP47 and CmP49) make up the majority of the haplotypes present in Southwest Indian DPS and appear to be ancestral haplotypes (based on presence in the South Atlantic and Southwest Pacific DPSs). The southwest Indian Ocean represents a genetic hotspot.
with 0.3–6.5 percent (mean=4.2 percent) estimated sequence divergence among the seven haplotypes identified. These haplotypes belong to three highly diverged genetic clades of haplotypes and highlights the complex colonization history of the region. There have been no nDNA studies from this region. There are no studies published on genetic stock composition at foraging areas within the Southwest Indian DPS.

8.2.5. Analysis of Factors Listed Under ESA Section 4(a)(1)

Section 4(a)(1) of the ESA lists five factors (threats) that must be considered when determining the status of a species. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

The following information on these factors/threats pertains to green turtles found in the Southwest Indian DPS.

8.2.5.1. Factor A: Destruction or Modification of Habitat or Range

Coastal development, beachfront lighting, erosion, sand extraction, consistently affect hatchlings and nesting turtles in portions of this DPS. The extent of sea grass and coral reef degradation is not known but are negatively affected by dredging and sedimentation and occurs in waters where green turtles are known to forage. All life stages of green turtles are affected by habitat destruction in the neritic/oceanic zone.

Terrestrial Zone

Habitat degradation is reported as an important source of additional mortality for this DPS, although the exact scale of habitat destruction at nesting beaches often is undocumented (Bourjea, 2012). In particular, habitat destruction, due to overdevelopment of the coastline and dredging or land-fill in foraging areas, is a threat to green turtles throughout Seychelles (Mortimer et al., 1996). Increase in tourism and human population growth on Mayotte Island, may lead to further negative impacts upon this coastal environment (Bourjea et al., 2007a). The possible negative effects of artificial lighting at a main nesting beach on Aldabra are of concern at the Seychelles (Mortimer et al., 2011) although currently being addressed (J. Mortimer; Seychelles Dept. of Environment, pers. comm., 2014) These factors may reduce the amount of available nesting area, and may evoke a change in the natural behaviors of adults and hatchlings (Ackerman, 1997).

Neritic Zone

In Mohéli, Comoros Islands, habitat degradation, due to sedimentation, sand extraction, and coral reef/seagrass bed degradation is also a concern (Ahamada, 2008). Similar situations are...
reported for Tanzania (Bourjea, 2012) and Madagascar (Ciccione et al., 2002; Rakotonirina and Cooke, 1994 as cited in Bourjea, 2012).

8.2.5.2. Factor B: Overutilization

Egg harvest occurs to a lesser extent than turtle harvest within this DPS. Turtles are harvested on the nesting beach and in foraging areas. The killing of nesting females continues to threaten the stability of green turtle populations in many areas affecting the DPS by reducing adult abundance and reducing egg production.

Egg Harvest

Egg poaching and/or hunting of nesting green turtles have been reported for Comoros Islands (Ahamada, 2008; Bourjea, 2012); Seychelles, especially in the inner islands where nesting populations are on the verge of local extinction (Mortimer, 2006; 2004 as cited in Bourjea, 2012; Mortimer et al., 1996); Mozambique (Costa et al., 2007; Videira et al., 2008); Tanzania (Bourjea, 2012); Madagascar (Ciccione et al. as cited in Bourjea, 2012; Lillette, 2006 as cited in Bourjea, 2012; Rakotonirina and Cooke, 1994); and Kenya (Bourjea, 2012). Egg harvest has affected green turtle populations in the Maldives (Seminoff et al., 2004). Illegal egg collection in Mauritius seems to be an important source of mortality but no data are available.

Turtle Harvest

Nesting green turtles at the Seychelles have increased at protected sites but declined where there has been heavy poaching, as on the developed islands of Mahé, Praslin and La Digue (Bourjea, 2012). On Assumption Island (9°45'S, 46°29'E), due to overharvesting, the number of nesting females decreased from over 5,000 estimated females in the early 1900s to under 250 females in the early 1980s. During 1982, more than 100 nesting turtles were killed by being turned overing on the nesting beach. A similar decrease has been reported for Aldabra, where green turtles were heavily exploited until the late 1960s1968, which is located just 20 miles from Assumption Island (Mortimer, 1984). After 1968; however, green turtles at have been protected at Aldabra (J. Mortimer, Seychelles Dept. of Environment, 2014) have been protected since 1968.

Areas of particularly heavy exploitation of green turtles include foraging locations in the Western Indian Ocean. One of the areas of greatest concern is in Madagascar. Fisheries take turtles as a target and as a bycatch. There is a long history of fishers taking sea turtles for meat and it continues today (Bourjea, 2012; Mbindo, 1996; Rakotonirina and Cooke, 1994). Direct capture of juvenile and adult turtles takes place using a variety of gear types in the traditional (artisanal) and industrial fisheries. Artisanal fishery, such as beach seining captures and entanglement in gill nets, has been reported to take tens of thousands of turtles annually (Hughes, 1981; Rakotonirina, 1987; Rakotonirina and Cooke, 1994; Lillette, 2006; Humber et al., 2010). This exploitation affects turtles nesting at Eparses Islands, where poaching and illegal trade at international foraging grounds are also a threat (Rakotonirina and Cooke, 1994; Lauret-Stepler et al., 2007). Similarly, commercial and small-scale fisheries at foraging grounds along the east African coast, mainly Tanzania and Kenya, affect green turtles nesting on Mayotte, Comoros.
Islands (Bourjea et al., 2007b). The Seychelles also continue to have ongoing intentional capture of green turtles (Seminoff et al., 2004) as does the east coast of Africa (Baldwin et al., 2003; Louro et al., 2006). Threats in South Africa are relatively well managed with a virtual absence of direct take. At Tanzania, bottom set gill nets pose a major threat to sea turtles. These mortalities are both incidental and targeted and while numbers vary, surveys suggest that 45-60 percent of gill net fishing trips catch turtles, accounting for several thousand turtles annually (not only green turtles). See also Section 8.2.5.5.1 (Incidental bycatch in fishing gear) below.

8.2.5.3. Factor C: Disease or Predation

The prevalence of FP in the Southwest Indian DPS is not known but it is the most deleterious disease among green turtle populations. FP is extremely rare among green turtles in Seychelles (J.A. Mortimer, unpublished data). It is found to affect juvenile to adult size turtles.

Nest and hatchling predation on the Southwest Indian DPS occurs although the level and extent of predation is not known for all areas of the DPS.

In the Southwest Indian DPS, this often-fatal disease has been reported for green turtle subpopulations in Kenya (Seminoff et al., 2004). However, despite these reports, there is no evidence that FP is widespread in the DPS.

Depredation occurs on green turtles found in the Southwest Indian DPS. Side striped jackals (Canis adustus) and honey badgers (Melivora capensis) are known to depredate nests (Baldwin et al., 2003) on the mainland coast of East Africa.

8.2.5.4. Factor D: Inadequacy of Existing Regulatory Mechanisms

There are a minimum of 15 national and international treaties and/or regulatory mechanisms that pertain to the Southwest DPS (see Conservation Efforts section). Hykle (2002) and Tiwari (2002) reviewed the value of some international instruments, which vary in their effectiveness. Often, international treaties 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, 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 a 2002 special issue of the Journal of International Wildlife Law and Policy: International Instruments and Marine Turtle Conservation (Hykle, 2002).
8.2.5.5. **Factor E: Other Natural or Manmade Factors**

The Southwest Indian DPS of the green turtle is negatively affected by both natural and manmade impacts. Within Factor E, we find that fishery bycatch that occurs throughout the DPS, particularly bycatch of green turtles from long lining operations, small prawn trawl fishery, and coastal gill nets can affect juvenile to adult size turtles. In addition, climate change and natural disasters are expected to be an increasing threat to all life stages of green turtles throughout this DPS.

**Incidental Bycatch in Fishing Gear**

Quantifying the magnitude of the threat of fisheries on green turtles in the Southwest Indian DPS is very difficult given the low level of observer coverage and investigations into bycatch conducted by countries that have large fishing fleets. As such, the full extent of the threat of incidental capture of sea turtles in artisanal and commercial fisheries in the Southwest Indian DPS is unknown. Sea turtles are caught in demersal and pelagic longlines, trawls, gill nets, and seines (Costa et al., 2007; Fennessy and Isaksen, 2007; Louro et al., 2006; Peterson, 2005; Peterson et al., 2007; 2009). There is evidence of significant historic bycatch from prawn fisheries, which may have depleted nesting populations long before nesting surveys were initiated in the 1990s (Baldwin et al., 2003). In the Seychelles, bycatch by local and international fisheries is a management concern, particularly by tuna long-liners and purse seiners (Mortimer et al., 1996), but its significance is unknown. Bycatch in small-scale and commercial fisheries along the east African coast (mainly Tanzania and Kenya) is a threat for turtles that nest on Mayotte and in the Comoros. (Bourjea et al., 2007a). Very few data on interactions with fisheries are available for Comoros. Domestic fisheries in this country are mainly composed of artisanal small mesh nets, unregulated and thought to be important source of mortality related to interaction with adult green turtles. Besides foreign tuna fisheries (purse seine and longline) that operate in the Exclusive Economic Zone (EEZ) of Comoros, there have been substantial shark fishing rights allocated to foreign countries. No data are available with respect to bycatch of turtles in these fisheries. Although very few data are available for fisheries activities in the French Eparses Islands, the offshore longline fishery of the French islands and the Seychelles seems to have a very small impact on sea turtles with very low incidental capture and mortality rates (Bourjea, 2012).

Interactions with a number of fisheries exist in the South Africa EEZ, notably with long lining operations, small prawn trawl fishery, and coastal gill nets. Information seems to indicate that the relative mortality due to fisheries either as targeted or incidental is approximately 95 percent of all documented turtle mortalities in Kenya (Wamukoya et al., 1997 as cited in Bourjea, 2012), with approximately 58 percent of sea turtles killed as a result of entrapment in fishing nets. Estimated incidental catch rates of turtles in shrimp trawls seems to be as high as 2–3 turtles/day during the shrimp season, equating to about 100-500 turtles/yr when TED were not in use.

**Climate Change, Natural Disasters, and Other**

Similar to other areas of the world, climate change and sea level rise have the potential to impact green turtles throughout the Southwest Indian Ocean. This includes beach erosion and loss from
rising sea levels, skewed hatchling sex ratios from rising beach temperatures, and abrupt disruption of ocean currents used for natural dispersal during the complex life cycle (Fish et al., 2005; Hawkes et al., 2009; Poloczanska et al., 2009). Climate change impacts could have profound long term impacts on nesting populations in the Southwest Indian DPS because much of the nesting occurs in low-lying islands and atolls; but it is not currently possible to quantify the potential. The pending sea level rise from climate change is also a potential problem, as this will inundate nesting sites and decrease available nesting habitat (Daniels et al., 1993). The French Eparses Islands and low-lying islands that are significant for green turtle nesting; sea level rise could affect hatching success of green turtles at those islands in the future.

Natural environmental events, such as cyclones, tsunamis and hurricanes, may affect green turtles in the Southwest Indian DPS. In general, however, severe storm events are episodic and, although they may affect green turtle hatchling production, their impacts are generally localized and they rarely result in whole-scale losses over multiple nesting seasons.

8.2.6. Summary of Existing Conservation Efforts

The Southwest Indian DPS is small but has experienced divergent nesting trends at different nesting sites. Although there is considerable uncertainty in anthropogenic mortalities, especially in the water, the DPS may have benefitted from important conservation efforts at the nesting beaches.

The international regulatory mechanisms described in Section 6.1.4 apply to green turtles found in the Southwest Indian DPS. In addition, green turtles of this DPS benefit from the Indian Ocean Southeast Asian Marine Turtle Memorandum of Understanding (IOSEA), and the Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region.


Also within the Southwestern Indian DPS, the Western Indian Ocean-Marine Turtle Task Force 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 Comment [A22]: This is a randomly organized list. I suggest you either list them in order by year of signing, or alphabetically by country name.

Response: this has been changed in the text to be chronological.
The Indian Ocean Tuna Commission (IOTC) is playing an increasingly constructive 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 on 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 in all long-liners operating in the Indian Ocean. (www.iotc.com).

8.2.6.1. Regional and National Legislation and Protection

In addition to these broad-reaching international instruments, there are several country-specific conservation efforts worth noting that occur within the Southwestern Indian Ocean. These are summarized below.

**Mozambique**

The nesting beaches in the Maputo Special Reserve (approximately 60 km of nesting beach) and in the Paradise Islands are within protected areas (Baldwin *et al.*, 2003; Costa *et al.*, 2007).

**Republic of Seychelles**

In the Seychelles Islands, the Turtle Act of 1925 protected only green turtles smaller than 30 inches in carapace length and focused more on ownership rights than on conservation (Mortimer *et al.*, 2011). Between 1945 and 1955, economic factors caused commercial exploitation at Aldabra to lapse temporarily, and between 1948 and 1962, a 6-month long closed season was established for female turtles at Aldabra (Mortimer, 1985). Since 1968, turtles at Aldabra have been well protected, but green turtles received little protection elsewhere in Seychelles until 1994 when 1994 Turtle Protection Regulations was implemented at the national level, making it illegal to kill any sea turtle or egg in Seychelles (Mortimer and Collie, 1998). In 1983, Aldabra became a UNESCO World Heritage Site managed by the Seychelles Islands Foundation. Since 1968, the human population at Aldabra has comprised only personnel directly employed on behalf of the Aldabra Research Station. Seychelles has plans to establish a network of outer island protected areas during the coming years which is likely to enhance protection at green turtle nesting beaches at other important nesting sites (Hays et al., in press).

**Comoros Islands**

The establishment of the Mohéli Marine Park in 2001 has been instrumental in the conservation of green turtles (Ahamada, 2008). Mohéli is the most important site of the Comoros Archipelago for green turtle nesting. The Mohéli Marine Park includes all the southern area from Miringoni Itsamia covering more than 40,000 ha. In addition, the Association pour le Développement Socio-Economique d’Itsamia has been preserving the importance of the beaches here for green turtles through protective actions since 1994 (Innocenti *et al.*, 2010).
Mayotte Island

The Directorate of Agriculture and Forestry teamed with the Department Organization of Mayotte to strengthen the protection of sea turtles in 1994. This was done by developing specific protection and conservation plans for the entire island, with special features for the two sites most frequented by green turtles (Bourjea et al., 2007a). In addition, the data collected on Mayotte (Roos et al., 2005; Taquet et al., 2006) show high abundance of foraging and nesting green turtles, and since monitoring started, data on nesting green turtles from other protected nesting sites in the southwest Indian Ocean have shown stability or significant increases. Mortimer (1985), Lauret-Stepler et al., 2007 and Bourjea et al. (2007a) suggest that this has been achieved through long-term conservation measures.

French Eparses Islands

There are six French islands scattered in the Southwest Indian Ocean in the vicinity of Madagascar. Three of the islands are important nesting sites for green turtles: Tromelin, Les Glorieuses and Europa (Lauret-Stepler et al., 2007). In 1971 they were all declared protected areas (DIREN, 2003).

8.2.6.2. International Instruments

Several regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the Southwest Indian DPS. The international instruments listed below apply to sea turtles found in this area and are described in Appendix 5.

- Convention on Biological Diversity
- Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention)
- Convention on the Conservation of Migratory Species of Wild Animals
- Convention on International Trade in Endangered Species of Wild Fauna and Flora
- Food and Agriculture Organization Technical Consultation on sea turtle-fishery interactions
- Indian Ocean Southeast Asian Marine Turtle Memorandum of Understanding
- Indian Ocean Tuna Commission
- Inter-American Convention for the Protection and Conservation of Sea Turtles
- International Convention for the Prevention of Pollution from Ships (MARPOL)
- International Union for the Conservation of Nature
- Memorandum of Understanding on ASEAN Sea Turtle Conservation and Protection
- Ramsar Convention on Wetlands
- United States Magnuson-Stevens Conservation and Management Act
8.3. Assessment of Significant Portion of its Range (SPR)

There are substantial threats to this DPS, including bycatch in fishing gear in nearshore and high seas regions, and high levels of turtle harvest along the coast of eastern Africa (e.g. Kenya and Somalia) and Madagascar, albeit reduced from former levels at some sites. The Southwest Indian DPS has four nesting sites indicating greater than 10,000 total nesters (i.e., estimate of total nesting females over 3 years). Satellite telemetry indicates that all four of the major nesting sites exhibit similar movement patterns, with a large number of animals from each nesting site moving to the eastern Africa/Madagascar coasts. Thus, the impacts of these threats are likely consistent across rookeries in this DPS. Based on long-term nesting data, the five most abundant nesting sites are increasing in annual nesting abundance, with the sixth largest (Tromelin Island, France) apparently showing a decreasing trend (1.7 percent/yr). However, the SRT concluded that, even if the Tromelin nesting site was lost, the DPS would not be at a substantially higher risk of extinction. Because the nature and degree of threats are relatively uniform across the range of the Southwest Indian DPS and loss of the one nesting site that is at greater risk of extinction would not result in a substantially higher risk of extinction to the DPS as a whole, the SRT concluded that the need to consider a significant portion of the range does not apply to this DPS.

8.4. Assessment of Extinction Risk

For the SRT's assessment of extinction risk for green turtles in the Southwest Indian DPS, there were two separate sets of ranking exercises: One focusing on the importance that each SRT member placed on each of the six different critical elements for this region (Table 8.3), and a second which reflects the SRT members’ expert opinion about the probability that green turtles would fall into any one of six different extinction probability ranges (Table 8.4). See Section 3.3 for details on the six elements and the voting process.

Table 8.3. Summary of ranks that reflect the importance placed by each SRT member on the critical assessment elements considered for the Southwest Indian DPS. For Elements 1-4, higher ranks indicate higher risk factors.

<table>
<thead>
<tr>
<th>Critical Assessment Elements</th>
<th>Element 1</th>
<th>Element 2</th>
<th>Element 3</th>
<th>Element 4</th>
<th>Element 5</th>
<th>Element 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abundance (1 to 5)</td>
<td>1.25</td>
<td>1.75</td>
<td>1.42</td>
<td>1.58</td>
<td>-0.75</td>
<td>0.75</td>
</tr>
<tr>
<td>Trends / Productivity (1 to 5)</td>
<td>0.13</td>
<td>0.13</td>
<td>0.15</td>
<td>0.15</td>
<td>0.22</td>
<td>0.18</td>
</tr>
<tr>
<td>Spatial Structure (1 to 5)</td>
<td>1-2</td>
<td>1-2</td>
<td>1-2</td>
<td>1-2</td>
<td>(-2)-0</td>
<td>0-2</td>
</tr>
<tr>
<td>Diversity / Resilience (1 to 5)</td>
<td>1-2</td>
<td>1-2</td>
<td>1-2</td>
<td>1-2</td>
<td>(-2)-0</td>
<td>0-2</td>
</tr>
<tr>
<td>Five-Factor Analyses (-2 to 0)</td>
<td>-0.75</td>
<td>0.22</td>
<td>-0.75</td>
<td>-0.75</td>
<td>0.75</td>
<td>0.75</td>
</tr>
<tr>
<td>Conservation Efforts (0 to 2)</td>
<td>0.75</td>
<td>0.18</td>
<td>0.75</td>
<td>0.75</td>
<td>0.75</td>
<td>0.75</td>
</tr>
</tbody>
</table>

With respect to the important rankings for the six critical assessment elements, all elements received low risk scores where the means were close to the minimum possible values. The large abundance of nesting females, increasing trends at main nesting beaches, large spatial
distributions, and various successful conservation measures were considered to help reduce the probability of extinction in the next 100 years. Variability among the SRT members was small, indicating the general agreement among the SRT members.

SRT members also generally thought that future threats not yet reflected in nester abundance or not yet experienced by the population weighed equally in their risk assessment voting to the conservation efforts are not yet reflected in nester abundance.

**Table 8.4.** Summary of Green Turtle SRT member expert opinion about the probability that the Southwest Indian DPS will reach a critical risk threshold within 100 years. Each SRT member assigned 100 points across the rank categories. This is a continuum with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1–5%</th>
<th>6–10%</th>
<th>11–20%</th>
<th>21–50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>MEAN ASSIGNED POINTS</strong></td>
<td>71.33</td>
<td>13.58</td>
<td>7.58</td>
<td>5.50</td>
<td>2.00</td>
<td>0.00</td>
</tr>
<tr>
<td><strong>SEM</strong></td>
<td>9.20</td>
<td>3.76</td>
<td>2.84</td>
<td>4.17</td>
<td>1.66</td>
<td>0.00</td>
</tr>
<tr>
<td><strong>Min</strong></td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td><strong>Max</strong></td>
<td>99</td>
<td>50</td>
<td>30</td>
<td>50</td>
<td>20</td>
<td>0</td>
</tr>
</tbody>
</table>

Of the six critical risk threshold categories describing the probability that the Southwest Indian DPS will reach a critical risk threshold within 100 years (Table 8.4), SRT member votes resulted in the greatest probability range of ‘<1%’ (mean=71.33). No vote was casted in the ‘>50%’ category.

In their vote justifications, most members cited the abundant females at wide-spread nesting beaches and the observed recent increasing trends at these nesting beaches. Some members noted the potential lack of enforcement along the east coast of the mainland Africa and possible negative effects on low-lying nesting beaches from climate change.

### 8.5. Synthesis and Integration

During the analysis of the Southwest Indian DPS’s status an integrated approach was taken by the SRT to consider the many critical elements described earlier. The Southwest Indian DPS is characterized by relatively high levels of green turtle nesting abundance and increasing trends.

The five-factor / threat analysis highlighted the continuing threats to the green turtle habitat that affects all life stages of green turtles. Nesting beaches throughout this DPS are susceptible to coastal development and associated beachfront lighting, erosion, and sea level rise. Nests and hatchlings are susceptible to predation, although the prevalence throughout the beaches of the Southwest Indian DPS is not known.
Coral reef and seagrass bed degradation continues in portions of the DPS affecting foraging turtles. Prevalence of FP within this DPS is unknown. Direct capture of juvenile and adult turtles continues to take place using a variety of gear types in the traditional (artisanal) and industrial fisheries.

The Southwest Indian DPS is protected by various international treaties and agreements as well as a few national laws. There are protected beaches throughout this DPS. As a result of these designations and agreements, many of the intentional impacts directed at sea turtles have been lessened, such as the harvest of eggs and adults in several nesting areas. The amount these threats are reduced as a result of these designations and agreements are not known.

The combined considerations of Abundance, Trends / Productivity, Spatial Structure, and Diversity / Resilience affected overall extinction risk threshold determinations. The Southwest Indian DPS was considered to have low risk of extinction in the next 100 years by the SRT. Although some threats to the DPS from fisheries interactions, direct harvest (eggs and adults), and climate changes exist, the observed recent increases at several nesting beaches within this DPS and large numbers of adult females, combined with successful conservation measures in this area, suggests that this DPS is unlikely to be extirpated in the near future. The DPS survived the intense harvest in the recent past and appears to be recovering steadily (Bourjea, 2012).
Section 9- North Indian

P1 1st paragraph: Israel and Jordan have been omitted from the list of countries in this DPS. They are only small sections of coast, but they exist.

Fig 9.1: there is now only 1 nesting beach/island in Kuwait (See Rees et al 2013 and Al-mohanna et al 2013) after the second site was destroyed by development. The second large ring in the Gulf seems to belong to Iran, and there is no evidence of any sizable nesting population existing there (Mobarakri articles and Pers Comm cited in text), the only green nesting in Iran is outside the Gulf.

Table 9.1: For a reviewer and possibly a critical reader it is annoying frustrating that one must go to an Appendix to see which references have been used to generate the values in the table and then elsewhere to check the reference and would be better served included in some form here. E.g. data for Kuwait; the Appendix indicated Rees et al 2012 (should be 2013) and Papathanasopoulou 2010 are used and yet Papathanasopoulou is not present in the reference list and the Rees et al 2012 reference present is for green turtles from Oman. Grobler et al 2001 (In Appendix 2) used for data in Oman is also not present in the reference list, neither is Al-Suweidi et al 2012 used in Appendix 2 for UAE. Thus, authors need to re-check all the data sources for similar oversight / errors / omissions.

The Kuwait – Qaru data presented should be from 2008 to 2011 not 2008 and 2011.

Fig 9.2: Do the years with zeros for Zabargad represent no nesting or no surveying... this needs to be made clear in the legend / on the chart.

Section 9.2.3: Omission that satellite tagging has been carried out in Kuwait. N=3 with one being a nester (Rees et al. 2013 CCB). References lacking for the other locations supplied. Egyptian tracks not yet published so need permission to be referred to.

“Satellite telemetry data for two post nesting females in Oman has not been analyzed nor published.” This is out of date. Rees et al 2012 CCB is referred to elsewhere in the text and appears in the reference list.

Al-Mohanna & George 2010 is cited. This is a paper on the genetics of a loggerhead in Kuwait and appears (incompletely) as such in the reference list. There is an Al-Mohanna & George abstract on green turtles in the Goa proceedings, could this be the correct reference?

It is unclear as to what information this large list of references that appears at the end of the section is supplying “(Minton, 1966; Ross, 1979; Ross and Barwani, 1982; Hirth, 1997; Pilcher and Al-Merghani, 2000; Ekanayake and Ranawana, 2001; Firdous, 2001; Venkatesan et al., 2004; Kapurusinghe et al., 2005; Sunderraj et al., 2006a; Ekanayake et al., 2006; Al-Mohanna
Section 9.2.5.1
Terrestrial zone: Second paragraph states the largest nesting population is Ras Al-Hadd, Oman but Table 9.1 indicates it is Ras Sharma in Yemen. “In the United Arab Emirates, increasing coastal development and associated light pollution, as well as vehicle ruts on the beaches, have been identified as threats to nesting habitat (Al-Abdessalaam et al., 2008)” - Only one green nest has been found (on an Island) in the UAE (http://www.seaturtle.org/mtn/archives/mtn133/mtn133p16.shtml) so reference to beach destruction in this country is irrelevant for this assessment unless ALL literature on beach conditions around the DPS is included.

Section 9.2.5.2
“Qatar (Tayab and Quiton, 2003; Pilcher, 2006)” - No green turtle nesting occurs in Qatar (it’s all hawksbill nesting) so reference to take of eggs in Qatar is incorrect.

Section 9.2.5.3
“In Qatar, depredation of eggs and hatchlings by foxes has been identified as a key source of turtle mortality (Al-Muraikhi et al., 2005; Pilcher, 2006). According to Pilcher (2006), more than 80 percent of nests within Ras Laffan Industrial City in Qatar were depredated by foxes during the 2005 season.” – again this refers to hawksbill nests as no green turtle nesting occurs in Qatar and so is irrelevant.

Section 9.2.5.5
“Beach driving has also been identified as a threat to sea turtles on beaches in Qatar (Tayab and Quiton, 2003; Pilcher, 2006) and the United Arab Emirates (Al-Abdessalaam et al., 2008).” - Both of these refer to hawksbill nesting beaches as greens do not nest in these countries and are therefore irrelevant.

Section 9.2.6
Oman “Oman banned bottom trawlers in 2009 (Andy Willson, Five Oceans LLC, pers. comm., 2013).” This could possibly be better referenced by the Governmental Decree: Ministerial Decree 20/2009 (http://faolex.fao.org/cgi-bin/faolex.exe?rec_id=082541&database=FAOLEX&search_type=link&table=result&lang=eng&format_name=@ER ALL)

UAE “In addition, fishing methods such as trawling and drift nets have also been banned in national waters.” I would like to see all methods listed, not “such as” and this needs referencing.

Section 9.3 Second Paragraph:
Should read “Almost 75% if the nesting occurs in a small...”. If India is specifically mentioned with 2.4% of the nesting, then why is Iran not mentioned with 2.6% and Saudi Arabia with 2.8? Revise this paragraph.

Section 9.4 Second Paragraph: again the text seems to be selective as it states: “the six Critical Elements, the largest range in rankings (i.e. voter opinion) is for the Diversity / Resilience Section (Table 9.3).” But Table 9.3 shows Element 2 (Trends) to have equal range as the stated Diversity / Resilience section. Revise this paragraph.

Section 9.5
This reads that “The North Indian DPS has a high level of green turtle nesting abundance with two of largest nesting assemblages of green turtles in the world nesting in Yemen and Oman “ I am would like to know these sites ranks in the world nesting assemblage list and how much smaller they are from the actual largest in the world... Reading this paragraph does not indicate whether these populations are missing the top by an order of magnitude or only hundreds or thousands of turtles. Please rephrase this paragraph to address this concern / point of clarification.
10. EAST INDIAN-WEST PACIFIC DPS (DPS #6)

10.1. DPS Range and Nesting Distribution

The boundary for the East Indian–West Pacific DPS is from 84°E longitude with the boundary in India near Odisha, northeast and into the West Pacific Ocean to include Taiwan extending to 142°E longitude, south to 4.5°N latitude then to West Papua in Indonesia and the Torres Straits in Australia, and to the southern extent of the range encompassing the Gulf of Carpentaria.

Green turtle nesting is widely dispersed throughout the East Indian-West Pacific DPS, with important nesting sites occurring in Northern Australia, Indonesia, Malaysia (Sabah and Sarawak Turtle Islands), Peninsular Malaysia, and the Philippine Turtle Islands (Figure 10.1). The largest nesting site lies within Northern Australia, which supports approximately 25,000 nesting females, calculated from the 5000 nesting female’s order of magnitude (Limpus, 2009). Currently, the East Indian-West Pacific DPS hosts 58 reported nesting sites (in some cases nesting sites are made up of multiple beaches based on nesting survey information) with six of these sites supporting more than 5,000 nesting females each (including the 25,000 nesters in Northern Australia). Nonetheless, populations are substantially depleted from historical levels.

Figure 10.1. Nesting distribution of green turtles in East Indian-West Pacific DPS (blue-shaded area). Size of circles indicates estimated nester abundance (see Section 10.2.1). Locations marked with ‘×’ indicate nesting sites lacking abundance information.
The in-water range of the East Indian-West Pacific DPS is similarly widespread with shared foraging sites throughout the DPS. Tagged green turtles that nest in western Australia have been resighted in Arnhem Land and as far north as the Java Sea near Indonesia (Baldwin et al., 2003; Limpus et al., 2007). The extensive coastline and islands of Indonesia support a large range of nesting and foraging habitat for green turtles (Halim and Dermawan, 1999). Waayers and Fitzpatrick (2013) found that in the Kimberley region of Australia, the green turtle appears to have a broader migration distribution and numerous potential foraging areas. A satellite-tagged female green turtle at Redang, Malaysia, travelled near Koh Samui, Thailand (Liew, 2002). Green turtle foraging grounds are known around the Andaman and Nicobar Islands (Andrews et al., 2006).

10.2. Critical Assessment Elements

In the evaluation of extinction risk for green turtles in the East Indian-West Pacific DPS, the SRT considered six distinct elements, each of which are discussed below: (1) Nesting Abundance, (2) Population Trends (3) Spatial Structure, (4) Diversity / Resilience, (5) Five-Factor Threat Analyses, and (6) Existing Conservation Efforts.

10.2.1. Nesting Abundance

There are 58 nesting sites (some sites are grouped by survey area and include multiple beaches) in the East Indian-West Pacific DPS for which data are available (Table 10.1). The largest nesting aggregation is found in Northern Australia, with an estimate of 25,000 nesting green turtles at Bountiful Island, Pisonia, and Rocky Islands near Mornington Island (Limpus, 2009). In Western Australia, Ningaloo hosts more than 6,000 nesters. In Indonesia, there are four main nesting areas known to host more than several hundred green turtle nests annually: Aceh (north Sumatra including Bangkaru, Belambangan Island), Pangumbahan (west Java), Berau Islands (east Kalimantan), and the Aru Islands (southwest Moluccas; Dethmers, 2010). The Sabah Turtle Islands in Malaysia and Baguian Island in the Philippines also maintain high concentrations of green turtle nesters although lower than historical levels. Other nesting sites in the Andaman and Nicobar Islands in India and Con Dao Island in Vietnam support more than 500 nesters. There are minor nesting sites in St. Martin’s Island, Bangladesh; Thameehla Island, Myanmar; Ishigaki Island, Japan; Taipin Tao, Taiwan; and Khram Island, Thailand.
Table 10.1. Summary of green turtle nesting sites in the East Indian-West Pacific DPS. Data are organized by country, nesting site, monitoring period, and estimated nester abundance \(\frac{\text{[Total Counted Females/Years of Monitoring]}}{\text{Remigration Interval}}\), and represent only those sites for which there were estimates of abundance. Many nesting sites, including relatively large ones, in the East Indian-West Pacific DPS are data deficient and estimates could not be made for those beaches. For a list of references for these data, see Appendix 2.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Australia</td>
<td>Ashmore Reef</td>
<td>1994 (3 weeks) and 1998 (2 weeks)</td>
<td>1</td>
</tr>
<tr>
<td>Australia</td>
<td>Barrow Island</td>
<td>1998–2004</td>
<td>Not calculated (974 crawls observed)</td>
</tr>
<tr>
<td>Australia</td>
<td>Bigge Island, Cassini Island, Cocos (Keeling) Islands, Hat Point, Jane Bay, Jurabi Coastal Park, Ningaloo MP, Lamarck Island, Maret Islands, Montalivet Island, Montebello Island, Muiron Islands, Red Bluff</td>
<td>1999</td>
<td>900</td>
</tr>
<tr>
<td>Australia</td>
<td>Cartier Island</td>
<td>1998</td>
<td>Not calculated (1 nest observed)</td>
</tr>
<tr>
<td>Australia</td>
<td>Cape Range NP</td>
<td>2008–2010 (40 days of monitoring)</td>
<td>30</td>
</tr>
<tr>
<td>Australia</td>
<td>Coral Bay</td>
<td>2008–2009 (34 days of monitoring)</td>
<td>7</td>
</tr>
<tr>
<td>Australia</td>
<td>Lacepedes Islands</td>
<td>2006</td>
<td>Not calculated (500–1,000 crawls observed)</td>
</tr>
<tr>
<td>Australia</td>
<td>Lowendal Island group</td>
<td>1998–2003</td>
<td>Not calculated (4 crawls observed)</td>
</tr>
<tr>
<td>Australia</td>
<td>Ningaloo, North West Cape</td>
<td>2009–2010</td>
<td>6,269</td>
</tr>
<tr>
<td>Northern Australia</td>
<td>Wellesley Group (3 sites of Bountiful Island, Pisonia and Rocky Islands near Mornington Island)</td>
<td>No survey year provided(^1)</td>
<td>25,000</td>
</tr>
<tr>
<td>COUNTRY</td>
<td>NESTING SITE</td>
<td>MONITORING PERIOD (YRS)</td>
<td>ESTIMATED NESTER ABUNDANCE</td>
</tr>
<tr>
<td>------------------------</td>
<td>--------------------------------------------------</td>
<td>-------------------------</td>
<td>----------------------------</td>
</tr>
<tr>
<td>Northern Australia</td>
<td>Northern Territory (Arnhem Wessel, Cobourg,</td>
<td>1991–2004</td>
<td>Not calculated</td>
</tr>
<tr>
<td></td>
<td>Groote, Groote Eyelant, Pellew, Tiwi, Cobourg,</td>
<td></td>
<td></td>
</tr>
<tr>
<td></td>
<td>and Cobourg Peninsula)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Brunei</td>
<td>Brunei</td>
<td>2004</td>
<td>Not calculated</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(13 crawls observed on 10</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>beaches)</td>
</tr>
<tr>
<td>India</td>
<td>Great Nicobar Island</td>
<td>1991</td>
<td>Not calculated</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(1 crawl observed)</td>
</tr>
<tr>
<td>India</td>
<td>Little Nicobar Island</td>
<td>1991</td>
<td>Not calculated</td>
</tr>
<tr>
<td></td>
<td></td>
<td></td>
<td>(1 crawl observed)</td>
</tr>
<tr>
<td>India</td>
<td>Andaman and Nicobar Islands, India</td>
<td>2001</td>
<td>750</td>
</tr>
<tr>
<td>Bangladesh</td>
<td>St. Martin Island</td>
<td>1996 to 2001</td>
<td>23</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Amandangan</td>
<td>2009</td>
<td>905</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Bangkaru</td>
<td>1999</td>
<td>62</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Belambangan Island</td>
<td>2000</td>
<td>1,736</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Bilang-Bilangan</td>
<td>2008–2009</td>
<td>7,156</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Derawan</td>
<td>2002–2006</td>
<td>29</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Enu</td>
<td>1997–1998</td>
<td>2,048</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Mataha</td>
<td>2008</td>
<td>1,652</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Pangumbahan</td>
<td>2010</td>
<td>5,199</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Sambit</td>
<td>1998–2000</td>
<td>555</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Sangalaki</td>
<td>2003–2009</td>
<td>2,740</td>
</tr>
<tr>
<td>Japan</td>
<td>Ibaruma Beach</td>
<td>1995–2003</td>
<td>181</td>
</tr>
<tr>
<td>Japan</td>
<td>Lejima</td>
<td>1995–1996</td>
<td>1</td>
</tr>
<tr>
<td>Japan</td>
<td>Ishigaki Island</td>
<td>1995–2003</td>
<td>56</td>
</tr>
<tr>
<td>Malaysia</td>
<td>Lankanayan Island</td>
<td>1999–2001</td>
<td>43</td>
</tr>
<tr>
<td>Malaysia</td>
<td>Mentawak</td>
<td>2011</td>
<td>11</td>
</tr>
<tr>
<td>Malaysia</td>
<td>Pahang</td>
<td>2002</td>
<td>188</td>
</tr>
<tr>
<td>Malaysia</td>
<td>Perak</td>
<td>No survey provided†</td>
<td>150</td>
</tr>
<tr>
<td>Malaysia</td>
<td>Redang Island</td>
<td>2004–2008</td>
<td>278</td>
</tr>
<tr>
<td>Malaysia</td>
<td>Sabah Turtle Island Park (Gulissaan Island,</td>
<td>2009–2011</td>
<td>7,011</td>
</tr>
<tr>
<td></td>
<td>BakkunaanKechil, Selinga Island)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Malaysia</td>
<td>Sarawak Turtle Island</td>
<td>1999–2001</td>
<td>1,155</td>
</tr>
<tr>
<td>Malaysia</td>
<td>Sipadan</td>
<td>2001</td>
<td>585</td>
</tr>
</tbody>
</table>

Comment [A1]: The best compilation for this is probably Andrews et al 2006 (attached). There is additional historical data from Satish Bhaskar’s surveys which we compiled into an article for Indian Ocean Turtle Newsletter 16 (see articles in this issues http://www.iotn.org/iotn-16.php, and attached PDF).

Response: Reviewed both papers, good information on individual estimates for various years. Used estimate for all islands to calculate nester abundance therefore kept 2001 reference.
<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Malaysia</td>
<td>Terranganu</td>
<td>1984–2000</td>
<td>1,875</td>
</tr>
<tr>
<td>Myanmar</td>
<td>KaingthaungKyun</td>
<td>1999</td>
<td>1</td>
</tr>
<tr>
<td>Myanmar</td>
<td>Thameehla Island</td>
<td>2007</td>
<td>109</td>
</tr>
<tr>
<td>Philippines</td>
<td>Panikian Island</td>
<td>2000</td>
<td>354</td>
</tr>
<tr>
<td>Philippines</td>
<td>Baguan Island</td>
<td>2004–2008</td>
<td>8,4325,874 (^1)</td>
</tr>
<tr>
<td>Philippines</td>
<td>Taganak</td>
<td>2000–2008</td>
<td>58963,7 (^2)</td>
</tr>
<tr>
<td>Philippines</td>
<td>Lihiman</td>
<td>2000–2008</td>
<td>8061,217 (^2)</td>
</tr>
<tr>
<td>Philippines</td>
<td>Langaan</td>
<td>1991–2008</td>
<td>6558808 (^3)</td>
</tr>
<tr>
<td>Philippines</td>
<td>Great Bakkunagun</td>
<td>2003–2006</td>
<td>118117,7 (^3)</td>
</tr>
<tr>
<td>Taiwan, Province of China</td>
<td>Lanyu</td>
<td>2010</td>
<td>19</td>
</tr>
<tr>
<td>Taiwan, Province of China</td>
<td>LiuChiu Island</td>
<td>2011</td>
<td>23</td>
</tr>
<tr>
<td>Taiwan, Province of China</td>
<td>Taipin Tao</td>
<td>2009</td>
<td>67</td>
</tr>
<tr>
<td>Taiwan, Province of China</td>
<td>Wan-an</td>
<td>2002</td>
<td>26</td>
</tr>
<tr>
<td>Thailand</td>
<td>Huyong Island</td>
<td>2004</td>
<td>105</td>
</tr>
<tr>
<td>Thailand</td>
<td>Khram Island, Sea Turtle Conservation Center of the Royal Thai Navy</td>
<td>2011</td>
<td>297</td>
</tr>
<tr>
<td>Thailand</td>
<td>Tarutao National Park</td>
<td>2003</td>
<td>1</td>
</tr>
<tr>
<td>Viet Nam</td>
<td>Con Dao Island</td>
<td>2001</td>
<td>900</td>
</tr>
<tr>
<td>Viet Nam</td>
<td>Minh Chau Island</td>
<td>2004</td>
<td>300</td>
</tr>
</tbody>
</table>

\(^1\) EPA Queensland Turtle Conservation Project unpublished data as Cited in Limpus(2009)

\(^2\) Cited in Liew (2002)

\(^3\) This number was updated based on external review subsequent to SRT voting, although we don’t believe the revisions would appreciably change how the SRT evaluated risk for the DPS.

Comment [A2]: New information from commenter.
Table 10.2. Green turtle nester abundance distribution among nesting sites in the East Indian–West Pacific DPS.

<table>
<thead>
<tr>
<th>NESTER ABUNDANCE</th>
<th># NESTING SITES</th>
</tr>
</thead>
<tbody>
<tr>
<td>n/a</td>
<td>7</td>
</tr>
<tr>
<td>1–10</td>
<td>7</td>
</tr>
<tr>
<td>11–50</td>
<td>8</td>
</tr>
<tr>
<td>51–100</td>
<td>4</td>
</tr>
<tr>
<td>101–500</td>
<td>11</td>
</tr>
<tr>
<td>501–1000</td>
<td>98</td>
</tr>
<tr>
<td>1001–5000</td>
<td>62</td>
</tr>
<tr>
<td>5001–10000</td>
<td>5</td>
</tr>
<tr>
<td>10001–100000</td>
<td>1</td>
</tr>
<tr>
<td>&gt;100,000</td>
<td>0</td>
</tr>
<tr>
<td>TOTAL SITES</td>
<td>58</td>
</tr>
<tr>
<td>TOTAL ABUNDANCE</td>
<td>7977,64009</td>
</tr>
<tr>
<td>PERCENTAGE at LARGEST NESTING SITE</td>
<td>33.2% (Wellesley Group, Australia)</td>
</tr>
</tbody>
</table>

10.2.2. Population Trends

Green turtle populations within the East Indian–West Pacific DPS have experienced apparent increases at some nesting sites, and decreases at others. For a list of references on trend data, see Appendix 3.

Information for the Suka Made (MeruBetiri National Park, East Java, Indonesia) suggests that nesting has declined since the early 1970s. Schulz (1987) reports a mean of approximately 1,500 nests per year from 1970–1974, which is substantially greater than the mean of 395 nests per year from 1991–1995 as reported by Arrinal (Limpus, pers. comm., 2002). At Pahgumbahan (West Java, Indonesia), the mean annual egg harvest was 2.5 million eggs in the 1950s and 400,000 eggs in the 1980s (Schulz, 1987). This apparent decline should be interpreted cautiously since it could be reflective of a decline in collection efforts rather than a decline in egg production. Likewise, at Thamihla Kyun, Maxwell (Groombridge and Luxmoore, 1989) reported a mean annual egg harvest of about 1.74 million eggs from 1883–1898, while in 1999, fewer than 250,000 eggs were harvested (Thorbjarnarson et al., 2000). Despite the apparent declines at Pahgumbahan and ThamihlaKyin, the lack of recent and/or multiple year datasets prevents an assessment of the current trends at these sites.
For Western Australia, there are four primary nesting concentrations, located at North West Cape, and on the islands of Lacepede, Scott Reef, and Ashmore Reef. Few data are available, although it has been estimated that the mean annual number of nests is somewhere between 3,000 and 30,000 (R. Prince, Dept. of Environment & Conservation, Bentley Delivery Center, pers. comm., 2001). The data are not sufficient to draw any conclusions regarding long-term trends in Western Australia. These sites, together with the Wellesley Group in Northern Australia, may constitute the most important green turtle nesting concentration in the Indian Ocean.

Nesting also occurs in many areas in the Southeast Asia region. These include Gulf of Thailand, Vietnam, Berau Islands and Enu Island (Indonesia), the Philippine Turtle Islands, and Sabah Turtle Islands, Sipadan, Sarawak, and Terengganu (all in Malaysia). Data suggest that populations have declined at the Gulf of Thailand, Vietnam, the Berau Islands, and perhaps Enu Island, although updated information is needed for these sites. The other sites in the Philippines and Malaysia appear to be stable.

Annual nesting in the Khram Island, Sea Turtle Conservation Center of the Royal Thai Navy, Gulf of Thailand has decreased from a mean of approximately 405 nests per year between 1975–1983 to a mean of approximately 250 nests per year from 1992–2001 (Charuchinda and Monanunsap, 1998; Charuchinda et al., 2002).

In Vietnam, the only site for which monitoring has occurred for an appreciable period is Con Dao National Park, monitored since 1995. Here, annual nesting of green turtles has remained relatively stable, with an annual mean from 1995–2003 of 239 females (World Wildlife Fund, unpublished data, and Nguyen Thi Dao, 1999 as cited in Hamann et al., 2006a). Outside of Con Dao, there appear to have been substantial decreases. For example, prior to the 1960s, approximately 500 females nested each year along the mainland beaches and near-shore islands of south-central Vietnam and approximately 100 females nested each year on islands in the Gulf of Tonkin (Hamann et al., 2006a). However, these breeding populations have declined significantly and probably number approximately 10 nests per year in both the Gulf of Tonkin and south-central Vietnam mainland coast (Hamann et al., 2006a).

In the Berau Islands (northeast Kalimantan, Indonesia), green turtle nesting has decreased over the last 60 years. Schulz (1984) estimated that approximately 36,000 females nested each season in the 1940s, with roughly 200 females per night during the peak of the nesting seasons. In the mid-1980s (the most recent data), approximately 4,000–5,000 females nested each season, with about 25 females per night during the peak nesting periods (Schulz, 1984). However, the data for the 1940s have not been verified and may be reflective of number of nests rather than females (N. Pilcher, Marine Research Foundation, pers. comm., 2007). This potential coupled with the lack of more recent data precludes trend analysis for this site.

Nesting beach monitoring has been ongoing sporadically at Enu Island (part of the Aru Islands in Indonesia) since the late 1970s (K. Dethmers, Australian Institute of Marine Science, pers. comm., 2007). There appears to have been a decline during these years, although the lack of continuous monitoring prevents an assessment of the current trend at this site. Nevertheless, data collected in 1997 (540 nesting females) suggest that this site remains an important nesting area
for green turtles in Southeast Asia (Dethmers, 2000; K. Dethmers, Australian Institute of Marine Science, unpubl. data).

At the Philippine and Sabah (Malaysia) Turtle Islands, both considered to be part of the same nesting population in the Sulu Sea (Moritz et al., 2002), information based on annual egg production and egg harvest indicates that nesting has remained stable in the Philippine Turtle Islands and may have increased at the Sabah Islands. In the Philippine Turtle Islands, egg production remained fairly stable from 1984–2000, with an annual mean of about 1.4 million eggs per year (Cruz, 2002). At Sabah, approximately 250,000 eggs were protected in the early 1980s (Groombridge and Luxmoore, 1989), a number that increased to nearly 1 million eggs by the late 1990s (E. Chan, Institute of Oceanography, KolejUniversetiSainsdanTeknologi, Malaysia, pers. comm., 2002), indicating an increasing trend. Although a mean of approximately 550,000 eggs were harvested annually from 1965–1968 (de Silva, 1982), these data represent eggs harvested, as opposed to those incubated or protected. Neither is reflective of total production, and Pilcher(2000) suggested that effort and data accuracy were dependable only after 1985.

At the Sipadan (Malaysia) nesting site, Chan (2006) reported that nesting levels have been fairly consistent each year from 1995 to 1999, numbering about 800 nests per year, with relatively little inter-annual variation in abundance.

In Sarawak and Terengganu (Malaysia), it appears that nesting abundance has been stable for 20 years or more. At Sarawak, approximately 2,000 nests were laid per year from 1970–2001, and at Terengganu, about 2,200 nests per year were laid from 1984–2000 (Liew, 2002; Chan, 2006). It should be noted, however, that data since 1927 (Banks, 1937) suggests that the current population, although stable, is dramatically reduced from historical levels.

Despite the numerous and widespread nesting beaches in this DPS, long-term monitoring data are relatively scarce. There are only two sites for which 10 or more years of recent data are available for annual nester abundance (one of the metrics for representing trends in this report), and these occur at two beaches in China, Wan-an and Lanyu, with estimated nester abundances of 26 and 19 respectively (Figure 10.2). There are four sites for which 15 or more years of recent data are available for annual nester abundance: Sabah Turtle Islands in Malaysia; Royal Navy Center in Khram Island, Thailand; Redang in Terrengganu, Myanmar; and Thameela Islands, Myanmar. See Section 3.2 for more on data quantity and quality standards used for trends and PVAs in this report. It should be noted that the nesting sites that met the standards of the SRT for plotting trends and conducting PVA do not represent anything near the majority of the nester population, so they do not represent the true status or trends in the DPS but simply provide information on the sites that have long-term data. While true trends cannot be ascertained in many cases due to the lack of data, we discuss the indications of possible trends at some of the primary nesting sites below.
As stated above, PVAs were conducted for nesting sites that had a minimum of 15 years of recent nesting data with an annual nesting level of more than 10 females (Figure 10.3; for more on data quantity and quality standards used, see Section 3.2). To assist in interpreting these PVAs, we indicate the probability of green turtle nesting populations declining to two different biological reference points, one using a trend-based and the other an abundance-based threshold. The trend-based reference point for evaluating population forecasts is half of the last observed abundance value, i.e., a 50 percent decline. The abundance-based reference point was a total adult female abundance of 300 females. Risk is calculated as the percentage of model runs that fall below these reference points after 100 years. For a full discussion of these reference points and PVAs, see Section 3.2.
Figure 10.3. Stochastic Exponential Growth (SEG) Model Output for the Sabah Turtle Islands and Redan, Terengganu, Malaysia; the Royal Navy Center, Thailand; and Thameehla Island, Myanmar. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference. Nesters were computed from nest counts using 4.5 nests per female.

This PVA has limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. The PVA indicates that the nesters from Sabah Turtle Islands in Malaysia, with an estimated 7,000 nesters, will likely continue to increase, while the nesters from the Royal Navy Center in Khram Island, Thailand (estimated 297 nesters), Redang in Terrengganu, Myanmar (estimated 278 nesters), and Thameela Islands, Myanmar (estimated 109 nesters) will likely continue to decline.
For the Sabah Turtle Islands in Malaysia, there is a 0.0 percent probability that this population will fall below the trend reference point (50% decline) at the end of 100 years. There is a 0.0 percent probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years. This trend may be a result of effective conservation measures made by the Sabah Government more than 20 years ago in the 1970s when the Sabah Turtle Islands were acquired from private ownership to provide complete protection to the nesting turtles and their eggs (Chan, 2006).

For the Royal Navy beaches in Thailand, there is a 100 percent probability that this population will fall below the trend reference point (50 percent decline) within 100 years. There is a 100 percent probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years.

For Redang in Terengganu, Malaysia, there is a 72.9 percent probability that this population will fall below the trend reference point (50 percent decline) at the end of 100 years. There is a 89.8 percent probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years.

For Thameela in Myanmar, there is an 87.9 percent probability that this population will fall below the trend reference point (50 percent decline) at the end of 100 years. There is a 96.7 percent probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years.

10.2.3. Spatial Structure

When examining spatial structure for the East Indian-West Pacific DPS, the SRT examined three lines of evidence: genetic data, flipper and satellite tagging, and demographic data. Genetic sampling in the East Indian-West Pacific DPS has been relatively extensive with more than 22 rookeries sampled. That said, there are many more rookeries in this region, there appears to be a complex population structure, and there are gaps relative to distribution (e.g., in Thailand, Vietnam, parts of Indonesia, and the Philippines). Overall, this region is dominated by a few common and widespread haplotypes and has varying levels of spatial structure characterized by the presence of rare/unique haplotypes at most rookeries. Within the DPS, there is significant population substructuring (pairwise FST 0.10–0.95, p<0.05). Of 22 rookeries studied, 16 regional genetic stocks have been identified in the East Indian-West Pacific DPS: Northwest Shelf, Scott Reef, Ashmore Reef, and the Gulf of Carpentaria (Australia); West Java, Berau Islands, and Aru (Indonesia); Peninsular Malaysia, Sarawak, Southeast Sabah (Malaysia), Sulu Sea (Malaysia/Philippines); Wan-an Island, and Lanyu Island (Taiwan); Zamami, Iriomote Island, and Ishigaki Island (Japan; Dethmers et al., 2006; Cheng et al., 2008; Hamabata et al., 2009; Nishizawa et al., 2011).

Tagging and tracking studies have been geared to studying inter-nesting migrations, and defining the range of inter-nesting habitats and post-nesting migrations. Green turtles that were satellite tracked from Pulau Redang, Terengganu indicate migrations to the South China Sea and Sulu Sea areas (Liew, 2002). Cheng (2000) reported movements of eight post-nesting green turtles from
Wan-An Island, Taiwan using Argos-linked satellite transmitters. The turtles distributed widely on the continental shelf to the east of mainland China. Destinations included southern Japan (Kyushu and Okinawa), Taiwan, and mainland China. Satellite telemetry studies conducted from 2000 to 2003 demonstrated that the green turtles nesting at Taipin Tao are a shared natural resource among the nations in the southern South China Sea. Green turtle females tracked in the same area travelled long distances commencing a post-nesting migration. Eleven green turtles tracked with satellite transmitters migrated in two directions: the first route stretched eastward along the eastern coast of the Gulf of Thailand to the Vietnam peninsula then some crossed the South China Sea and entered Sulu Sea in the Philippines; the second route went south across the Gulf of Thailand to the Malaysia peninsula travelling a distance that ranged from 456 to 2,823 km (Charuchinda et al., 2002) to the China Sea and the remaining one migrated north to the coastal region of Japan (Wang, 2006). Waayers and Fitzpatrick (2013) found that in the Kimberly region of Australia, the green turtle appears to have a broad migration distribution and numerous potential foraging areas.

Mixed stock analysis of foraging grounds shows that green turtles from multiple nesting beach origins commonly mix at feeding areas in foraging grounds across northern Australia (Dethmers et al., 2010) and Malaysia (Jensen, 2010) with higher contributions from nearby large rookeries. There is evidence of low frequency contribution from rookeries outside the DPS at some foraging areas.

The demography of green turtles in the East Indian-West Pacific DPS varies throughout the nesting assemblages. This variation in parameters such as mean nesting size, remigration interval, inter-nesting interval, clutch size, hatching success, and clutch frequency suggests a high level of population structuring in this DPS. The size of nesters throughout the DPS range from 82.1cm CCL to 103.6cm CCL (Trono, 1991; Hirth, 1997; Charuchinda and Monanunsap, 1998; Basintal, 2002). Growth rates: 0.83 cm/yr for nesting females (Pilcher and Basintal, 2000). Clutch size varies among rookeries from 87.2 to 115 eggs per nest. Remigration interval also varies from 2 to 5 years, and clutch frequency from 1.67 to 8 nests per season. Hatching success ranges from 37 percent to 94 percent with some sites in incubation facilities (Hendrickson, 1958; Suwelo, 1971; Trono, 1991; Leh, 1994; Hirth, 1997; Abe et al., 1998, 2003; Charuchinda and Monanunsap, 1998; Pilcher and Basintal, 2000; Tiwol and Cabanban, 2000; Basintal, 2002; Chan et al., 2007; Kobayashi et al., 2008; Adnyana et al., 2008; Zainudin et al., 2008; Lwin, 2009a, 2009b; Cheng et al., 2009; Jensen, 2010; Waayers, 2010; Chen et al., 2010; Dethmers, 2010; Muhara and Herlina, 2012; Reischig et al., 2012).

10.2.4. Diversity / Resilience

The components considered under this critical element include the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation (e.g., high vs. low beach face, insular vs. continental nesting sites), and the genetic diversity within the DPS. Components such as these are important considerations for assessing the potential impact of events and phenomena such as storms, sea level rise, and disease. Nesting and foraging areas are widespread within this DPS, providing a level of population resilience through habitat diversity.
The nesting season varies throughout the DPS, with nesting from June to August in the inner Gulf of Thailand and on Derawan Island the peak nesting season occurs from March to July (Charuchinda and Monanunsap, 1998; Abe et al., 2003; Aureggi et al., 2004; Adnyana et al., 2008). Green turtles nest year-round in Aru, Indonesia, and in Thameela Island in Myanmar (Lwin, 2009a; Dethmers, 2010). The peak nesting season occurs from November to March in Aru, Indonesia (Dethmers, 2010), Sukamade, southeastern Java (Arinal, 1997), Barrow Island, and western Australia (Pendoley, 2005).

Nesting occurs on both insular and continental sites, yielding a degree of nesting diversity. Limited information also suggests that there are two types of nesters within the DPS, those with high site fidelity which nest regularly at one site, such as the Sabah Turtle Islands, and those with low site fidelity such as at Ishigaki Island (Basintal, 2002; Abe et al., 2003).

### 10.2.5. Analysis of Factors Listed Under ESA Section 4(a)(1)

Section 4(a)(1) of the ESA lists five factors (threats) that must be considered when determining the status of a species. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

#### 10.2.5.1. Factor A: Destruction or Modification of Habitat or Range

Coastal development, beachfront lighting, erosion resulting from sand mining, and sea level rise, as a result of climate change, consistently affect hatchlings and nesting turtles throughout this DPS extending to protected beaches. Driving on beaches is a threat in some areas, such as Australia. The extent of fishing practices, depleted seagrass beds, and marine pollution is broad with high levels occurring in waters where high numbers of green turtles are known to forage and migrate. All life stages of green turtles are affected by habitat destruction in the neritic/oceanic zone.

**Terrestrial Zone**

In the East Indian-West Pacific DPS, the majority of green turtle nesting beaches are extensively eroded. Nesting habitat is degraded due to a variety of human activities largely related to tourism. Coastal development and associated artificial lighting, sand mining, and marine debris affect the amount and quality of habitat that is available to nesting green turtles. However, there are sanctuaries and parks throughout the region where nests are protected to various degrees.

Comment [A3]: There are numerous major ports along this coast now, but it is not clear what their impacts are or are going to be. Response: May not know all impacts but do believe it would obstruct littoral drift as in Komar (1983).
mining and coastal development as a result of tourism related activities are the main threats to the nesting habitat (Fatima et al., 2011). In 2004, a major earthquake occurred off the west coast of Sumatra, Indonesia, resulting in significant tsunami waves. This tsunami permanently altered large stretches of beach, particularly in the Nicobar Islands. Post-tsunami surveys of nesting beaches in the Andaman Islands showed reformed beaches with some areas showing signs of nesting (Andrews et al., 2006). Many green turtle nesting beaches in the Andaman Islands were not significantly affected.

Current nesting populations in Bangladesh are small, with fewer than 30 nests recorded during a 6-year period (1996–2001) on St. Martin’s Island (Islam, 2002). The beaches in Bangladesh are under threat from coastal development stemming from the tourist industry. Alterations of sand dunes and nesting beaches are recognized as a major threat to sea turtles in Bangladesh (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). Three important nesting beaches were declared Ecologically Critical Areas in 1999: Sonadia Island, Cox’s Bazar to Teknaf peninsular coast, and St. Martin Island (Rashid, 2006). The following activities are banned within these three Ecologically Critical Areas: Wildlife killing; turtle collecting, including shell collection; industry and structure establishment; pollution of the soil and water; and any activity that threatens the natural state of the land and water. However, a deep sea port built at the northern end of Sonadia Island threatens nesting habitat (Islam et al., 2011).

In Myanmar, green turtle nesting primarily occurs on the beaches of Thameehla Island (Diamond Island), an island at the mouth of the Pathein River (Thorbjarnarson et al., 2000). The island is protected year round by the Department of Fisheries. However, these nesting beaches are susceptible to flooding and sand loss (Lwin, 2009a).

Green turtle nesting habitat in Thailand is threatened by coastal development (Settle, 1995). The Khram Islands, an important nesting site for green turtles in Thailand, has been under the protection of the Royal Thai Navy (Charuchinda and Monanunsap, 1998). Thailand coastal areas in the Provinces of PhangNga, Krabi and Phuket sustained extensive damage as a result of the 2004 tsunami (Aureggi and Adulyanukosol, 2006). The extent of the damage to green turtle foraging habitat is unknown.

In Malaysia, destruction and modification of green turtle nesting habitat has resulted from coastal development and construction for tourism. Nesting habitat is degraded and permanently lost by activities such as beach nourishment and construction of sea walls and jetties (Chan, 2004). Coastal development for tourism on Sipadan in Sabah, Malaysia, has degraded the nesting habitat on this beach (Palaniappan et al., 2004). Turtle sanctuaries have been established in Terengganu (RantauAbang Turtle Sanctuary; Ma’Daerah Turtle Sanctuary; PasirTemit, Hulu Terengganu; PasirLubokKawah, Hulu Terengganu; PasirKumpal, Dungun), Perek (PantaiJabatan, Perak River), Sabah (Turtle Islands Park), Sarawak (Talang-Satang National Park), and three beaches in Redang Island. Coastal development continues to threaten all other nesting beaches (Chan, 2004, 2006, 2010).
The beaches in Indonesia are being lost due to erosion from high tides and monsoons. Sangalaki Island in Indonesia is one of the largest known green turtle nesting sites in the Celebes Sea. Extensive logging activities on Borneo have caused an increase in drift wood on the nesting beach. These logs make parts of the beach inaccessible to nesting turtles (Obermeier, 2002).

In Vietnam, green turtle nesting has declined in all areas except on the protected beach in Con Dao National Park where nesting numbers are stable (The Ministry of Fisheries, 2003). Most of the beaches in Vietnam 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 (The Ministry of Fisheries, 2003), and sand mining operations on Minh Chau and Quan Lan also threaten nesting habitat.

Historically, green turtle nests were found along the coast of China from Fujian to the Beibu Gulf, as well as islands in the South China Sea. These nesting sites have been degraded due to tourism-related coastal development and sand mining. These developed beaches suffer from light pollution as well as tourists’ activities on the nesting beaches. As a result, many beaches along the mainland coast and offshore islands no longer host green turtle nesting (Chan et al., 2007), and green turtle nesting habitat only occurs at a few remaining sites. Huidong Gangkou Sea Turtle National Nature Reserve, set up in 1986, is one of the remaining nesting sites for green turtles in China (Wang, 2006).

In Taiwan, the beaches at Wan-an (Wangan) Island are important nesting sites for green turtles. These beaches have been designated as a green turtle refuge. This refuge was established in 1995 and provides some protection to both turtles and the nesting beach (Tan, 2004). However, this area is also an important tourist destination, and the impacts of tourism and tourism-related infrastructure may still pose a problem for the nesting beach habitat (Cheng, 1995). This nesting habitat has also been affected by sand mining and coastal development (Cheng et al., 2009). On Lanyu (Orchid Island), Taiwan, street lighting along the coastal highway near Badai beach, where most of the nesting on the island occurs, creates unsuitable nesting habitat and causes hatchling disorientations (Cheng et al., 2009).

In Australia, the majority of green turtle nesting along the beaches of the Gulf of Carpentaria occurs outside of the protection of the National Park. Other minor nesting sites lie within Indigenous Protected Areas (IPA). These lands are protected from development (Limpus, 2009).

In Western Australia, the impacts to nesting and hatchling green turtles by independent turtle watchers as well as off-road vehicles has increased in the Ningaloo region as the number of visitors has increased over the years (Waayers, 2010). Nesting turtles and hatchlings are routinely disturbed by people with their cars and flashlights (Kelliher et al., 2011). The operation of motor vehicles on the beach affects sea turtle nesting by interrupting or striking female turtles on the beach, headlights disorienting or misorienting emergent hatchlings, vehicles running over hatchlings attempting to reach the ocean, and vehicle tracks interfering with hatchlings crawling to the ocean. Hughes and Caines (1994) found that loggerhead hatchlings appeared to become diverted not because they cannot physically climb out of the rut, but because the sides of the track cast a shadow and the hatchlings lose their line of sight to the ocean horizon.
(Mann, 1977). The extended period of travel required to negotiate tire tracks and ruts may also increase the susceptibility of hatchlings to dehydration and depredation during migration to the ocean (Hosier et al., 1981). Driving on the beach can cause sand compaction which may result in adverse impacts on nest site selection, digging behavior, clutch viability, and emergence of hatchlings, decreasing nest success, and directly killing pre-emergent hatchlings (Mann, 1977; Nelson et al., 1987; Nelson, 1988; Limpus, 2002). Physical changes and loss of plant cover caused by vehicles on dunes can lead to various degrees of instability, and therefore encourage dune migration. As vehicles move either up or down a slope, sand is displaced downward, lowering the trail. The possible effects of disturbance by wildlife tourism-related activities can be seen from the low nesting at Jurabi Coastal Park compared to number of nesting turtles at the Nangaloo Marine Park in Western Australia (Waayers, 2010).

Burn-off flares associated with oil and gas production on the Northwest shelf of Australia are in sufficiently close proximity to the green turtle nesting beaches to possibly cause hatchling disorientation. At Thevenard Island, the risk of hatchling disorientation due to these flares is greater (Pendoley, 2000)

**Neritic/Oceanic Zones**

The loss and degradation of seagrass habitat is an issue of great concern for green turtles. A global study of seagrass losses found that the Indo-Pacific region has the largest gaps in information regarding seagrass coverage and trends, which is especially problematic given the rapid human population growth and development in that region (Waycott et al., 2009).

Green turtles forage in the seagrass beds around the Andaman and Nicobar Islands in India. Some of these seagrass beds in the South Andaman group are no longer viable foraging habitat due to siltation and degradation due to waste disposal, a byproduct of the rapid increase in tourism (Andrew, 2000). Little is known about the foraging areas in the middle and north Andaman Islands or the Nicobar Islands. Green turtles that forage off the waters of the Bay of Bengal in south Bangladesh also face depleted foraging habitat from divers collecting seagrass for commercial purposes. Seagrass in the Bay of Bengal is also being degraded by the anchoring of commercial ships, ferries, and boats in this habitat (Sarkar, 2001). In the nearshore waters of Thailand, seagrass beds are partially protected since fishing gear such as pouch nets and trawls are prohibited (Charuchinda et al., 2002). In the waters surrounding the islands of Togean and Banggai in Indonesia, the use of dynamite and potassium cyanide are common, and this type of fishing method destroys green turtle foraging habitat (Surjadi and Anwar, 2001).

Seagrass beds are found throughout the nearshore areas of Vietnam’s mainland coast and islands (The Ministry of Fisheries, 2003; Nguyen Van Tien et al., 2002 as cited in The Ministry of Fisheries, 2003; Vo Si Tuan, 2002 as cited in The Ministry of Fisheries, 2003). Destructive fishing practices have been and possibly continue to be a major threat to this habitat in 21 of Vietnam’s 29 provinces (Asia Development Bank, 1999 as cited in the Ministry of Fisheries, 2003). Although these destructive fishing practices are prohibited by legislation passed in 1989, enforcement may not be sufficient to prevent these practices from occurring. Green turtle foraging habitat is under increased threat from decreased water quality through river run-off and development (The Ministry of Fisheries, 2003).
Destructive fishing practices also degrade green turtle foraging habitat in the waters of Indonesia. In 1991, an area including six uninhabited islands was declared the Aru Tenggara Marine Reserve; however, green turtles are not specifically included as one of the turtles to be in protected in this Reserve (Dethmers, 2010). Destructive fishing practices, including cyanide and dynamite fishing, also occur in the waters of the Turtle Islands of Indonesia (Cruz, 2002), which threaten green turtle foraging habitat.

In Malaysia, loss of feeding grounds for green turtles occurs due to nutrient run-off, sedimentation, and pollution including contaminants and debris (National Research Council, 1990c; Chan, 2004).

In 1999, the waters surrounding the Turtle Islands in the Philippines, 15 km from the shoreline of each island in the Southwestern Sulu Sea, were declared a protected area known as the Turtle Islands Wildlife Sanctuary, pursuant to Republic Act, Presidential Proclamation No. 171 (Cruz, 2002). While this provides some protection to seagrass beds in these waters, destructive fishing practices, including cyanide and dynamite fishing, still occur in the waters of the Turtle Islands of the Philippines (Cruz, 2002), which threaten green turtle foraging habitat.

10.2.5.2. Factor B: Overutilization

Overutilization for commercial and subsistence purposes likely was a factor that contributed to the historical decline of this DPS. Current harvest of green turtles for human consumption continues on a portion of this DPS and affects all life stages from eggs to adults.

Egg Harvest

The green turtle populations within this DPS have mostly decreased throughout their range. Populations throughout Asia have been depleted through long-term harvests of eggs and adults, and as by-catch in the ever-growing fisheries (Shanker and Pilcher, 2003).

Despite substantial declines in green turtle nesting numbers, egg harvest remains legal in several of the countries within this DPS. In Bangladesh, green turtle nesting was common on most of these beaches. In 1989, 35 green turtles were recorded nesting in one night on one beach in St. Martin, Bangladesh (Islam, 2002). Egg collection is considered the most serious threat for the few green turtle nests in Bangladesh if not relocated to a hatchery (Islam et al., 1999 as cited in Islam, 2001, 2002). Over-exploitation has brought the nesting turtles to near extinction (Hasan, 2009).

Turtle eggs were historically collected and sold to visitors from the mainland of Myanmar, with about 1.6 million green turtle eggs harvested annually in the early 1900s (Thorbjarnarson et al., 2000; Islam, 2002). Prior to 1986, virtually all eggs were collected. From 1986 to 1996, one-third of the eggs were required to be left to hatch. After 1997, the Myanmar Fisheries Department fully protected all beaches where turtle nesting still occurred (Thorbjarnarson et al., 2000), and collection of eggs and harvest of nesting females were banned. However, sea
turtle eggs and nesting females continue to be harvested due to a lack of law enforcement (Islam, 2001).

In Thailand, the major threat to sea turtles is the exploitation of eggs and turtles for meat and turtle products. Egg collection continues on remote beaches that are not regularly monitored (Charuchinda et al., 2002). In Myanmar and Thailand, hatcheries are set up to protect a portion of the eggs. However, these hatcheries retain hatchlings for several days for tourism purposes, thus reducing the likelihood of hatchling survival (Charuchinda et al., 2002).

In the 1950s, the green turtle nesting population in Malaysia started showing a decline after decades of egg collection (Chan, 2006). In the early 1970s, less than 10 percent of eggs were retained for incubation in hatcheries in peninsular Malaysia. Over 4,100,000 eggs were harvested in Sarawak between 1967 and 1978, of which only 2 percent were transplanted to hatcheries. Green turtle eggs were nearly completely harvested in Sarawak right up until the 1980s. In 2001, the percentage of eggs protected in peninsular Malaysia increased to approximately 50 percent; the remainder was marketed (Siow and Moll, 1982). Not surprisingly, turtle nesting numbers began to decrease in peninsular Malaysia where the number of eggs laid in Terengganu went from 928,900 in 1956 to between 107,135 and 417,981 annually from 1984 to 1989 (Mortimer, 1992). In Sabah, from 1965 to 1978, a total of over 6,000,000 eggs were collected, and approximately 2,700,000 were transplanted to hatcheries (Siow and Moll, 1982).

After 40 years of intense egg harvest in Sabah, the nesting population declined (de Silva, 1982; Limpus, 1995). It was believed this decline could be attributed to egg harvesting, although turtles were also threatened from incidental capture in fishing gear (Mortimer, 1991). In order to provide some protection for turtles, all three Sabah Turtle Islands were acquired and protected by the Sabah State Government in the 1970s (de Silva, 1982). Egg collection dropped to approximately 250,000 in the early 1980s (Groombridge and Luxmoore, 1989), but had increased to nearly 1 million eggs by the late 1990s (E. Chan, Institute of Oceanography, Kolej Universiti Sains dan Teknologi, Malaysia, pers. comm., 2002 as cited in NMFS and FWS, 2007). Despite the protections on the three Sabah Turtle Islands, the nesting population continued to decline until 1987 when there were signs of some recovery for green turtles (Pilcher, 2000). However, after more than 20 years of conservation efforts (1970–1990), the population had still not shown signs of recovery (Limpus et al., 2001).

At Pahgumbahan in West Java, Indonesia, the mean annual egg harvest was 2.5 million eggs in the 1950s and 400,000 eggs in the 1980s (Schulz, 1987). However, this apparent decline could be reflective of a decline in egg collection efforts rather than a decline in egg production. Egg harvesting in Indonesia occurred for decades till 1999. Illegal egg harvesting continues, but there is an increased effort to fully protect green turtles from harvest on the islands of Bilang-Bilangan and Mataha in Indonesia (Reischig et al., 2012).

There are a few beaches in Malaysia, Indonesia, and Thailand where eggs are protected in hatcheries. However, in Sabah, Malaysia, hatcheries have been found to produce 100 percent females, which will skew green turtle sex ratios in the wild (Tiwol and Cabanban, 2000).
In addition to the harvest for meat, eggs were also harvested throughout the Indonesian archipelago with many as 2 million eggs taken off the beaches every year (Limpus, 1997).

In the Turtle Islands, owned by both the Philippines and Malaysia, an 88 percent decline in egg production between 1959 and 1992 can be attributed to the almost complete exploitation of all the eggs. However, the collection of eggs is now regulated and of an estimated egg production of 9,022,553 eggs between 1984 and 1992, 65 percent were conserved (Hirth, 1997). From 1984 to 2000, 71 percent of the 21,678,109 eggs laid in the Tawi-Tawi province of the Philippines were conserved, while 21 percent of the eggs were collected for consumption (Cruz, 2002).

Egg harvest remains a problem in Vietnam and was a principal factor in the decline of turtles nesting in that nation. Because nesting has declined so dramatically, egg harvest has also declined and become scattered and inconsistent. Although sea turtle nests are currently protected on Con Dao National Park and Nui Chua beaches in Vietnam, in unprotected areas nearly 100 percent of eggs are harvested (The Ministry of Fisheries, 2003). Green turtle nests (less than 10) laid on the Vietnam beaches along the Gulf of Tonkin have been reported as being susceptible to collection (Hamann et al., 2006a). Because of the decline of turtles in Vietnam, the number of turtles caught for consumption has also decreased; however, captures have been reported to continue at a low rate in seven coastal communities where at least one family catches turtles (Hamann et al., 2005).

In Japan, egg collection was common in the coastal areas during times of hunger and later by those who acquired them on the black market (Kamezaki et al., 2003) but is no longer a problem (Abe et al., 2003; Kamezaki et al., 2003). Currently, egg poaching in Japan is illegal due in large part to research and conservation efforts throughout the country. Laws were enacted in 1973 to prohibit egg collection on Yakushima, and in 1988, the laws were extended to the entire Kagoshima Prefecture (Matsuzawa, 2006).

**Turtle Harvest**

Nesting females continue to be killed in countries within Southeast Asia and the Indian Ocean (Fleming, 2001; Fretey, 2001; Cruz, 2002). In the 1800s, turtles and turtle eggs were an important food source for the indigenous people of the Andaman and Nicobar Islands in India (Shanker and Andrews, 2004). In 1977, a ban on hunting and harvesting of turtles came into force in the Andaman Islands, and sea turtles were protected under Schedule 1 of the Indian Wildlife (Protection) Act (1972) (http://envfor.nic.in/legis/wildlife/wildlife1.html); however, indigenous peoples of the Andaman and Nicobar Islands are exempt from the Indian Wildlife Act (Andrews et al., 2006). Egg and turtle harvest remains at a subsistence level.

In Bangladesh, since the 1980s, green turtle nesting populations have declined due to severe exploitation of eggs and illegal killing of adult turtles (Islam, 2002).

Indonesia has a lengthy history of exporting sea turtle products continuing to the 1990s (Mlleken, 1987; Groombridge and Luxmoore, 1989). Local islanders in Indonesia have traditionally considered turtles, especially green turtles, as part of their diet (Hitipeuw and Pet-Soede, 2004 as cited in FAO, 2004). About 25,000 green turtles were being exploited for meat
each year toward the end of the 20th century (Dethmers, 2010). In addition, in the 1960s and 1970s, Indonesia exported 25,000 to 50,000 stuffed turtles annually with the green turtle being the most common turtle.

Green turtles can be found in the waters and nesting along the beaches of the Kai islands in Indonesia. They are harvested whenever encountered to be used as meat. Suárez (2000, as cited in Limpus, 2009) recorded 173 green turtles captured with nets or hooks in the water or taken on the nesting beach during a 6-month period. The green turtle populations that formerly nested on Bali have been extirpated (Schulz, 1984; Groombridge and Luxmoore, 1989), but thousands of green turtles were being brought into Bali each year (21,000 in 1990), where they were butchered for meat for personal consumption and for Balinese ceremonies and rituals (Barr, 2001). In 1990 the Balinese government decreed that green turtle utilization would be limited to a maximum of 5,000 turtles per year, though actual numbers may be more than 50 percent higher (Halim et al., 2001). Available evidence indicates that egg and turtle harvests (on the nesting beach and at sea) are far in excess of sustainable levels (Groombridge and Luxmoore, 1989; Barr, 2001).

Turtle fisheries continue around Aru primarily for trade in Bali. Drift nets are set near the nesting beach and seagrass beds catching an average of 15 turtles per night (Dethmers, 2010). On the main islands within the Aru archipelago, where green turtles come to nest, the inhabitants are dependent solely on marine resources (Dethmers, 2010). Many nesting turtles are collected in the waters just off the Indonesian beaches where some fishermen collect as many as 300 turtles on a trip. This type of harvest extends out to Aru, Southeast Sulawesi, East Kalimantan, Irian Jaya, Madura, Timor and Flores. About 25 percent of the harvested turtles are males, which confirms that in addition to the harvest of nesting females on the beach, harvest also occurs at foraging and courtship grounds.

Mostly in the remote areas of the Philippines, green turtles are still killed and sold for meat, and eggs continue to be exploited for consumption and trading. The Pawikan Conservation Project was created in 1979 to address the decline of sea turtles in the Philippines and has been effective in promoting conservation and scientific management of the turtle resources although much still needs to be done (Ramirez-de Veyra, 1994). In addition to egg collection, green turtles are being incidentally taken during fishing activities in the waters surrounding the Turtle Islands, and the number of turtles taken is increasing with the increasing number of fishing vessels, particularly during illegal fishing operations by Chinese vessels. In 2002 alone, four vessels from China were caught with more than 58 turtles onboard, mostly green turtles (Cruz, 2002). Thus, despite increased conservation efforts, the killing of turtles and selling of turtle meat still occurs in the Philippines, mostly in remote areas (Cruz, 2002). Nearshore fisheries that incidentally take sea turtles in Taiwan and retain them for consumption and trade are also considered a threat to green turtles foraging in these waters (Cheng et al., 2009).

Despite legal protections for sea turtles, at-sea poaching of turtles is a continuing problem in Southeast Asia, especially by Hainanese and Vietnamese vessels. The poaching occurs in a wide-ranging area of the region, and has moved as turtle stocks have been depleted, with vessels being apprehended off Malaysia, Indonesia, and the Philippines (Pilcher et al., 2009 as cited in Lam et al., 2011). The apprehension of Chinese vessels with large numbers of sea turtles (tens
to hundreds), including green turtles, throughout the eastern Indian Ocean and South China Sea (Lam et al., 2011) highlights the problem, though it likely represents only a small portion of the poaching that occurs. It is notable that many of the fishermen that have been apprehended are aware of the laws and associated penalties for harvesting marine turtles, but do so under the cover of darkness and other times when they are aware that enforcement is limited (Lam et al., 2011).

Licensed fishermen in Japan can legally catch sea turtles for local consumption (Horikoshi et al., 1994). The annual number of turtles caught is 150. Of these turtles the majority are immature green turtles caught in the Yaeyama Islands (Abe et al., 2003).

In Australia, green turtles are harvested by Aboriginal and Torres Strait Islanders for subsistence purposes. Tens of thousands of turtles were harvested by indigenous people in the Ningaloo Region of Australia from the 1950s to the early 1970s (Limpus, 2002). The total annual harvest in Australian waters in the 1970s was estimated to be between 7,500 and 10,500 (Kowarsky, 1982). The most common method of capturing turtles is by harpoon from a boat. However, today there is a widespread use of motorized aluminum boats in contrast to the traditional dugout canoes powered by paddles or sail. Daly (1990) reported an estimate of 10,000 adult green turtles being harvested in the Torres Strait with about 4,000 of these taken by Torres Strait islanders and about 6,000 by Papua New Guineans for sale in their coastal markets (Hirth and Rohovit, 1992). In 2001, Morris and Lapwood recorded 96 green turtles were harvested on the Dampier Peninsula over a 4 month period, the majority adult-sized females, and estimated 500 green turtles harvested annually (K. Morris, pers. comm. as cited in Limpus, 2009). The total harvest of green turtles by indigenous people across northern and Western Australia is probably several thousand annually (Kowarsky, 1982; Henry and Lyle, 2003 as cited in Limpus, 2009). The indigenous harvest of eggs may be unsustainable in northeast Arnhem Land (Kennett et al., 1998).

### 10.2.5.3. Factor C: Disease or Predation

The presence of FP in green turtles occurs throughout this DPS, although the prevalence is not known. It may be an emerging threat brought about by human-related habitat degradation.

FP has been found in green turtles in Indonesia (Adnyana et al., 1997), Japan (Y. Matsuzawa, Japanese Sea Turtle Association, pers. comm., 2004), the Philippines (Nalo-Ochona, 2000), Western Australia (Raidal and Prince, 1996; Aguirre and Lutz, 2004), and on PhuQuoc in Vietnam (The Ministry of Fisheries, 2003). Epidemiological studies indicate rising incidence of this disease (George, 1997), thus the above list will likely grow in the future.

External visible tumors, most likely FP, have been reported by local turtle hunters foraging near the Wellesley Islands, Gulf of Carpentaria (EPA Queensland Turtle Conservation Project, unpublished data) and in the waters near the western coast of Australia (Raidal and Prince, 1996).

The best available data suggest that current nest and hatchling predation on the East Indian-West Pacific DPS is prevalent and can be an increasing threat without nest protection and predatory control programs in place. Depredation of nests by feral animals is also widespread in many South Asian areas (Sunderraj et al., 2001; Islam, 2002). Nest predation by feral pigs and dogs is
a major threat on the Andaman and Nicobar Islands of India (Fatima et al., 2011). Monitor lizards are also a significant and widespread predator in some areas (Andrews et al., 2006). Dog predation is a major threat to the green turtle nests on Sonadia Island in Bangladesh (Islam et al., 2011). Jackals, foxes, wild boars, and monitor lizards also predate green turtle nests and hatchlings along the beaches of Bangladesh, and dogs also kill or injure nesting females in Bangladesh (Andrews et al., 2006). Lizards and ghost crabs are the natural predators of green turtle nests in Thailand (Chantrapornsyl, 1993). In Malaysia, crabs (Ocypode spp.) predate green turtle eggs (Ali and Ibrahim, 2000), and gold-ringed cat snakes or mangrove snakes (Boigadendrophila), (Asiatic) reticulated pythons (Python reticulatus), monitor lizards (Varanus sp.), and house mice (Mus musculus) predate hatchlings (Hendrickson, 1958). Monitor lizards, crabs, and ants predate eggs and hatchlings on the beaches of Vietnam (as cited in “Sea Turtle Migration-Tracking & Coastal Habitat Education Program– An Educator’s Guide” http://www.ioseturtles.org/Education/seaturtlebooklet.pdf). In Japan, raccoon dogs (Nyctereutes procyonoides) and weasels (Mustela latiata) are a threat to nests (Kamezaki et al., 2003). In Taiwan, snakes predate the nests (Cheng et al., 2009).

Hendrickson (1958) estimated that 4 percent of the adult females on Malaysian beaches showed signs of assumed shark attack-amputated flippers and missing shell. It has been speculated that sharks congregate in large numbers around the Sarawak Turtle Islands during the peak breeding season (Hirth, 1997).

On the North West Cape and the beaches of the Ningaloo coast of mainland Australia, a long established feral European red fox (Vulpes vulpes) population historically preyed heavily on eggs and is thought to be responsible for the lower numbers of nesting turtles on the mainland beaches (Baldwin et al., 2003; Kelliher et al., 2011). During the 2010–2011 nesting season, foxes predated 23 percent of all the nests laid along the North West Cape and Cape Range Division of the Ningaloo Coast (Kelliher et al., 2011).

10.2.5.4. Factor D: Inadequacy of Existing Regulatory Mechanisms

Our review of regulatory mechanisms under Factor D demonstrates that although regulatory mechanisms are in place that should address direct and incidental take of East Indian-West Pacific DPS green turtles, these regulatory mechanisms are insufficient or are not being implemented effectively. In areas throughout the DPS, the killing and poaching of green turtles have been banned, however, due to the lack of awareness and implementation, turtles continue to be harvested. We find that the threat from the inadequacy of existing regulatory mechanisms for fishery bycatch (Factor E) and impacts to nesting beach habitat (Factor A) is a continuing threat throughout this DPS.

In addition to local and national regulatory mechanisms, there are a minimum of 16 national and international treaties and/or regulatory mechanisms that pertain to the East Indian-West Pacific DPS. Hykle (2002) and Tiwari (2002) have reviewed the effectiveness of some international instruments. The problems with existing international treaties are often that they have not realized their full potential, do not include some key countries, do not specifically address sea turtle conservation, and are handicapped by the lack of a sovereign authority to enforce environmental regulations. The ineffectiveness of international treaties and national legislation
is oftentimes due to the lack of motivation or obligation by countries to implement and enforce them. A thorough discussion of this topic is available in a special 2002 issue of the Journal of International Wildlife Law and Policy: International Instruments and Marine Turtle Conservation (Hykle, 2002).

There are beaches and in water habitat throughout the DPS that are under various levels of protection. The level of protection for green turtles depends on clear regulations and consistent funding for enforcement. Often the designation is not sufficient to protect sea turtles from being harvested. For examples, in 1991, the Aru Tenggara Marine Reserve declared as a strict marine reserve by governmental decree, an area of 114,000 hectares and including six uninhabited islands. The area is not demarcated, which makes it unclear where protective regulations apply. While loggerheads and leatherbacks receive protection in this Reserve, the decree does not include green turtles (Dethmers, 2010).

Fishery bycatch occurs throughout the East Indian-West Pacific DPS (see Factor E), as well as anthropogenic threats to nesting beaches and foraging grounds (Factor A) and eggs/turtles and foraging (Factors A, B, C, and E), is substantial. Although conservation efforts to protect some nesting beaches are underway, more widespread and consistent protection is needed. Although national and international governmental and non-governmental entities in the East Indian-West Pacific DPS are currently working toward reducing green turtle bycatch, as well as egg and turtle harvest, it is unlikely that this source of mortality can be sufficiently reduced across the range of the DPS in the near future. This is due to the lack of bycatch reduction in commercial and artisanal fisheries operating within the range of this DPS, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and lack of availability of comprehensive bycatch reduction technologies.

10.2.5.5. Factor E: Other Natural or Manmade Factors

The East Indian-West Pacific DPS of the green turtle is negatively affected by both natural and anthropogenic factors as described above. Fishery bycatch, particularly from drift net and purse seine fisheries, occurs throughout the East Indian-West Pacific DPS, with localized high levels of mortality in waters where juvenile to adult turtles are known to forage and migrate. In addition, vessel collisions, marine pollution, changes likely to result from climate change and natural disasters are also an increasing threat to all life stages of green turtles throughout this DPS.

**Incidental Bycatch in Fishing Gear**

Incidental capture in artisanal and commercial fisheries is a significant threat to the survival of green turtles in the East Indian-West Pacific DPS. Green turtles may be caught in drift and set gill nets, bottom and mid-water trawling, fishing dredges, pound nets and weirs, and haul and purse seines. While a comprehensive, quantitative assessment of the impacts of the East Indian-West Pacific DPS drift net fishery on turtles is impossible, it is likely that the mortality inflicted by the drift net fisheries in 1990 and in prior years was significant (Wetherall et al., 1993), and the effects may still be evident in sea turtle populations today. The high mortality of juveniles, subadults, and reproductive adults in the high-seas drift net fishery has probably altered the
current age structure (especially if certain age groups were more vulnerable to drift net fisheries) and therefore diminished or limited the reproductive potential of affected sea turtle populations.

Gill nets and set bag nets are the two major fishing gears used in the Bay of Bengal, and green turtles are likely captured during these fishing operations. Along the coast of Andaman and Nicobar Islands, the main type of fishery is gill nets and purse seines with thousands of turtles killed annually by fisheries operations including the shark fishery (Shanker and Pilcher, 2003; Chandi et al., 2012). Shark fishing was identified as one of the most serious threats to the green turtle population foraging in the waters off the coast of the Andaman and Nicobar Islands. In 1994, Bhaskar estimated at least 600 green turtles were killed as a result of the shark fishery in this area. Over the last decade, there has been an increase in the large predator fishing industry. Green turtle mortality can be expected to be much higher than that estimated in the 1990s as a result of these current operations (Namboothri et al., 2012).

Bangladesh fish for their livelihood using gill nets, set bag nets, trawl nets, seine nets, hook and line and other net types of gear (Hossain and Hoq, 2010), which are known to capture turtles. Trawl fishing is also common in Bangladesh. No green turtle stranding information is available to determine the fishery threat level to the green turtle population; however, it is expected to be high as TEDs are not used and the population has declined (Admedet et al., 2003).

Bycatch in fisheries using gears such as trawlers, drift nets, purse seines is thought to be one of the main causes of decline in the turtle population in Thailand. The rapid expansion of fishing operations is largely responsible for the increase in adult turtle mortality due to bycatch (Settle, 1995). The most used fishing gears in the waters of Thailand are trawling and drift gill nets. Heavy fishing is the main threat to foraging sea turtles (Chantrapornsyl, 1993).

In Cambodia, sting ray hook lines have caught sea turtles as bycatch. This type of fishing is now banned (Sereywath, 2006).

In Malaysia, fishing gears, such as drift nets, trawls nets, and purse seines, have been documented as a threat to green turtles (Liew, 2002). It is estimated that 245 and 100 green turtles, respectively, were incidentally caught in 1984 and 1985 in trawl nets and drift nets or gill nets off Terengganu, Malaysia (Chan et al., 1988).

Incidental capture of green turtles by net-based fisheries is the largest threat to the species in Vietnam. Green turtles are also caught opportunistically by divers seeking other commercial species such as mollusks or crustaceans (Hamann et al., 2006a). As a result of the unregulated fishery industry in Vietnam, there are approximately 4,000 sea turtle deaths per year; a portion of these deaths are green turtles (Norman et al., 2004).

In Indonesia, green turtles were recorded as one of the main species caught in the longline fisheries. Longline fisheries have an impact on green turtles in the East Indian-West Pacific DPS, although they are typically considered more of a concern for other turtle species in the region, especially olive ridleys. Fishery observers recorded 85 total turtle captures by the tuna longline fleet out of Bitung-North Sulawesi in Indonesia in May-December 2006 (832,208 hooks observed), with six of them being green turtles (Zainudin et al., 2008). Pocket bottom trawl gear
is allowed in eastern Indonesia waters. During 2-hour trawl operation times, 2-20 sea turtles were incidentally caught (Hitipeuw et al., 2006). Trawl gear is still allowed in the Arafura Sea, posing a major threat to green turtles (Dethmers, 2010). Shrimp trawl captures in Indonesia are high because of the limited use of Turtle Excluder Devices (TEDs) (Zainudin et al., 2008). Opportunistic capture of green turtles by divers seeking other commercial species such as mollusks or crustaceans also occurs in Indonesia (Dethmers, 2010).

On the Turtle Islands in the Philippines, there have been an increased number of dead turtles as a result of fishing vessels such as shrimp, trawlers, and demersal nets (Cruz, 2002). Most of these vessels come from Sabah, Malaysia, and Manila.

The estimated bycatch of the Japanese large-mesh drift net fishery in the North Pacific Ocean in 1990–1991 was 1,501 turtles, of which 248 were estimated to be green turtles (Wetherall et al., 1993). Wetherall et al. (1993) speculate that the actual mortality of sea turtles taken in the Japanese and Taiwanese large-mesh fisheries may have been between 2,500 and 9,000 per year.

**Pollution and debris**

Direct or indirect disposal of anthropogenic debris introduces potentially lethal materials into green turtle foraging habitats.

A study on the chemical contamination of green turtle eggs in peninsular Malaysia revealed persistent organic pollutant levels that were high, but not as high as in loggerheads from the same area, and not likely to result in sex reversal within the eggs (Ikonomopoulou et al., 2009). However, heavy metal concentrations were high enough to indicate an increased risk of embryonic development problems and reduced hatching success. Organic pollutants in green turtle hatchlings may compromise their offshore predator avoidance ability (van de Merwe et al., 2010).

Pollution from oil spills, as well as agro and organic chemicals, is a major threat to the waters used by green turtles in the Bay of Bengal (Sarkar, 2001). Berger, (1991) examined the potential environmental impact of offshore oil spills in the vicinity of Palawan, Philippines. The oil spill trajectories would be dependent upon spill location and time of the year. Depending on these circumstances, the green turtle nesting sites on Palawan and the Calamian Islands would be at risk. The result of the population growth in China has been an increased amount of pollutants in the coastal system. Discharges from untreated sewage have occurred in Xisha Archipelago Li et al. as cited in(Chan et al., 2007).

Concentrations of nine heavy metals (iron, manganese, zinc, copper, lead, nickel, cadmium, cobalt, and mercury) and other trace elements were found in liver, kidney, and muscle tissues of green turtles collected from Yaeyama Islands, Okinawa, Japan (Anan et al., 2001). The accumulation of cadmium found in the green turtles is likely due to accumulations of this heavy metal in the plant materials on which they forage (Sakai et al., 2000).

In the Gulf of Carpentaria, Australia, discarded fishing nets have been found to cause a high number of turtle deaths with the majority being green turtles (Chatto et al., 1995).
**Climate Change**

Similar to other areas of the world, climate change and sea level rise have the potential to impact green turtles in the East Indian-West Pacific DPS. Potential impacts include beach erosion and loss of nesting habitat from rising sea levels, skewed hatchling sex ratios from rising beach incubation temperatures, and abrupt disruption of ocean currents used for natural dispersal during the complex life cycle.

Extreme sand temperatures at nesting beaches also create highly skewed female sex ratios of hatchlings or threaten the health of hatchlings (NMFS and FWS, 1998; Fuentes et al., 2010). 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 hatchlings. As temperatures increase, there is also concern that incubation temperatures will reach levels that exceed the thermal tolerance for embryonic development, thus increasing embryo and hatchling mortality (Fuller et al., 2010b).

**Natural Disasters**

Natural environmental events, such as cyclones and hurricanes, may affect green turtles in the East Indian-West Pacific DPS. Typhoons have been shown to cause severe beach erosion and negatively affect hatching success at green turtle nesting beaches in Japan, especially in areas already prone to erosion. For example, Matsuzawa (2006) found that for loggerheads nests during the 2004 season, the Japanese archipelago suffered a record number of typhoons, and many nests were drowned or washed out. Without human intervention to protect clutches against these natural environmental threats, many of the nests in Japan would be lost. In general, however, severe storm events are episodic and, although they may affect green turtle hatching production, the results are generally localized and they rarely result in whole-scale losses over multiple nesting seasons.

**10.2.6. Summary of Existing Conservation Efforts**

There are numerous ongoing conservation efforts in this region; however, the level of anthropogenic mortalities remains high for the East Indian-West Pacific DPS, based on the best available information. Hatcheries have been set up throughout the region to protect a portion of the eggs laid and prevent complete egg harvesting, bycatch reduction efforts have been made in some areas, protected areas are established throughout the region, and monitoring, outreach and enforcement efforts have made progress in sea turtle conservation. Despite these conservation efforts, considerable uncertainty in the status of this DPS lies with inadequate efforts to measure bycatch in the region, a short time-series of monitoring on nesting beaches, and missing vital rates data necessary for population assessments.

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 Shankar and Andrews, 2004). Recently, a consolidated manual has been produced to achieve these goals by Dakshin Foundation and Madras Crocodile Bank Trust (MCBT) (Sea Turtles of India, 2011).

The 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 programme on Little Andaman Island. A multi-institution stakeholder platform for marine conservation, including government and non-government agencies, was established by these groups to facilitate the conservation of marine turtles and other endangered species (Tripathy et al., 2012).

Despite management plans and guidelines to protect sea turtle nest, protection hinges on regular monitoring and patrolling. This has been difficult due to the remoteness of the islands, lack of staff, and equipment to carry out the protection measures. Protection and hatchery practices, by the Forest Department, have occurred regularly on several key beaches in the Andaman Islands, Ramnagar beach; Cuthbert Bay beach and Rutland Island. During the 2000–2001 nesting season, hatchery practices were stopped for that season on Smith Island and Cuthbert Bay beach and in-situ nest protection was adopted.

In 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 TIHPA is one of the world's major nesting grounds for green turtles. Management of the TIHPA is shared by both countries. The implementing agencies include the TIHPA, the Pawikan Conservation Project under the Protected Areas and Wildlife Bureau of the Philippines Department of Environment and Natural Resources, and Sabah Parks of 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). It is not known the level of effectiveness and progress of these goals.

One of the nesting beaches for this DPS, Australia’s Dirk Hartog Island, is part of the Shark Bay World Heritage Area and recently became part of Australia’s National Park System. This designation may facilitate monitoring of nesting beaches and enforcement of prohibitions on direct take of green turtles and their eggs. Conservation efforts on nesting beaches have included invasive predator control. On the North West Cape and the beaches of the Ningaloo coast of mainland Australia, a long established feral European red fox (Vulpes vulpes) population preys heavily on eggs and is thought to be responsible for the lower numbers of nesting turtles on the mainland beaches (Baldwin et al., 2003). Fox populations have been eradicated on Dirk Hartog Island and Murion Islands (Baldwin et al., 2003), and threat abatement plans have been implemented for the control of foxes (1999) and feral pigs (2005).
Turtle parts for illegal trade continue to be a problem in the East Indian-West Pacific DPS. In order to reduce this threat, the Vietnamese Government, with assistance from IUCN, WWF, TRAFFIC and the Danish Government, formulated a Marine Turtle Conservation Action Plan in 2010 to expand awareness to fishers, and enforcement officers and confiscate sea turtle products (MoFI, 2004 as cited in Stiles, 2009). The level of effectiveness and progress of this program is not known.

TED use spread to Thailand, Malaysia, the Philippines, Indonesia and Brunei, by the initiatives of the South East Asian Fisheries Development Center (Food and Agriculture Organization of the United Nations, 2004). In 2000, the use of TEDs in the Northern Australian Prawn Fishery (NPF) was made mandatory. Prior to the use of TEDs in this fishery, the NPF annually took between 5,000 and 6,000 sea turtles as bycatch, with a mortality rate estimated to be 40 percent (Poiner and Harris, 1996). Since the mandatory use of TEDs has been in effect, the annual bycatch of sea turtles in the NPF has dropped to less than 200 sea turtles per year, with a mortality rate of approximately 22 percent (based on recent years). Initial progress has been made to measure the threat of incidental capture of sea turtles in other artisanal and commercial fisheries in the Southeast Indo-Pacific Ocean (Lewison et al., 2004; Limpus, 2009); however, the data remain inadequate for population assessment.


10.2.6.1. National Legislation and Protection

In addition to the international mechanisms, most East Indian-West Pacific countries have developed legislation to protect sea turtles and/or nesting habitats. They are summarized below. However, the overall effectiveness of this country specific legislation is unknown at this time.

Australia

Sea turtles in Australia are protected under the Environment Protection and Biodiversity Conservation Act of 1999 (EPBC Act), which implements several international agreements or conventions to which Australia is a signatory. The EPBC Act lists all marine turtles in Australia as ‘threatened’ species, and provides several mechanisms to address declines in population numbers of listed species that include: recovery plans, threat abatement plans, wildlife conservation plans, conservation agreements, and conservation orders. Traditional Owners, as recognized under the Australian Government’s Native Title Act of 1993, are able to assert their rights to gain customary authority for shared resources such as marine turtles which includes traditional hunting rights. In Western Australia, The Dampier Archipelago, Thevenard Island and Barrow Island, Ningaloo Marine Park, and Montebello Conservation Park are Nature Reserves. These Reserves protect green turtle nesting habitat (Limpus, 2009).
**Bangladesh**

The following legislation is relevant to green turtles in Bangladesh: National Environmental Conservation Act (1995); New Fisheries Management Policy, Bangladesh (1986); and Revised Bangladesh Wildlife (Preservation) Amendment Act that includes sea turtles (Islam *et al.*, 2011).

**Brunei Darussalam**


**Cambodia**

No national legislation exists to protect sea turtles (Shanker, 2004).

**China**


**Hong Kong**

In Hong Kong, all sea turtle species are protected under local laws: the Protection of Endangered Species of Animals and Plants Ordinance and the Wild Animals Protection Ordinance. Under the Protection of Endangered Species of Animals and Plants Ordinance, it is an offense to import, export, or possess any part of a sea turtle or its eggs. To effectively protect the nesting turtle and its habitat, the nesting beach of Sham Wan has been designated as a Restricted Area under the Wild Animals Protection Ordinance, which forbids entry during the green turtle nesting season from June to October.

**India**

All species of sea turtles are protected under Schedule 1 of the Indian Wildlife Protection Act of 1972, which provides legal protection to turtles from capture on nesting beaches and in coastal waters, as well as from trade. In the Andaman Islands, a ban on hunting and harvesting of turtles came into force in 1977. However, indigenous groups of people, the original inhabitants of the Andaman and Nicobar Islands, are still exempt from the Indian Wildlife Protection Act (Shanker and Andrews, 2004). National legislation has not been effective due to the lack of consultation and cooperation at the community level [Upadhyay and Upadhyay, 2002].

Comment [A7]: There are other legislations that are pertinent to sea turtle conservation, particularly habitat protection.

Response: Added Biodiversity Act.
The Biodiversity Act of 2000 identifies areas of high biodiversity such as sea turtle nesting beaches as “heritage sites” and includes measures to manage these sites (Upadhyay and Upadhyay, 2006).

Indonesia

In Indonesia, sea turtles are protected under a variety of decrees, acts, and regulations. Act No. 4 (1982) provides the basic legal provision for the management of the living environment, augmented by Act No. 5, which deals with the conservation of living natural resources and the environment. While other sea turtle species had some protection under other decrees, it was not until 1999 that green turtles were protected under Government Regulation No. 7, which provided protections to all sea turtle species (Zainudin et al., 2008). Green turtles became listed as a protected species under the Government Regulations 7/1999 and 8/19999.

Japan

In Japan, there are eight laws and ordinances that regulate (allow via permit) or prohibit actions harmful to sea turtles, such as taking, buying, and selling turtles, their eggs, and any derivative products, or restrict access to nesting beaches. The Law for the Conservation of Endangered Species of Wild Fauna and Flora is the primary law in Japan that intends to conserve endangered species. It prohibits the capture of sea turtles and eggs for sale for all seven species and prohibits domestic assignment or transfer of endangered species listed in CITES (UmigameHogo no tameno, 2006 as cited in Mason et al., 2010). This law was established in accordance with CITES and is enforced by the Japan Ministry of Environment (Maison et al., 2010).

Myanmar

Myanmar Marine Fisheries Law (1990) prohibits any kind of mechanized fishing within 5 miles of the coast (Win and Lwin, 2012). Regulations issued in 2005 by the Ministry of Fisheries prohibit the eating of turtle meat and eggs and require that turtles caught as bycatch in fishing nets be released, and trawlers must be equipped with devices to minimize the risk of turtle capture (Hamann et al., 2006b).

Thailand

Sea turtles were listed as protected species in 1947. The killing of sea turtles and the collection of eggs was prohibited. In 1972, the Fisheries Act prohibited commercial fishing within 3 km of the coastline. In 1979, the Ministry of Commerce Enactment prohibited the export of sea turtles. In 1992, the Conservation and Protection of Living Resources Enactment (Act No. 19) prohibited the collection, production, or sale of sea turtle products. In 1997, the use of TEDs in shrimp trawl fisheries was enforced (Charuchinda et al., 2002).

Malaysia

Malaysia has various wildlife protection acts and ordinances, as well as fishery regulations that include measures aimed at protecting sea turtles. They also have specific sea turtle protection
regulations in the form of the 1951 Turtle Enactment and two later amendments (1987, 1989 for Sabah; Shanker, 2004). The 1990 Regulation: Prohibition of Methods of Fishing bans the use of drift nets or gill nets with mesh sizes of more than 10 inches. The 1991 Regulation: The Fisheries Regulations 1991 declares waters off the coast of Merchang to Kampung Kuala Abang (Tanjung Jara, Trengganu) as a prohibited area.

**Philippines**

Executive Order 542 (1979) established the Task Force Pawikan, which enforces Ministry of Natural Resources Administrative Order No. 33 and No. 8, regarding marine turtle sanctuaries and the harvesting and exploitation of eggs in the Turtle Islands and Tawi-Tawi. The Philippines also has a 1999 Presidential Proclamation that established the Turtle Islands Wildlife Sanctuary, and the Wildlife Act of 2001 that provides for conservation and protection of wildlife resources (including marine turtles) and their habitats (Shanker, 2004).

**Taiwan**

The green turtle has been classified as an endangered species in the Taiwan's Wildlife Conservation Law (promulgated on June 23, 1989) and amended in 2009.

**Vietnam**


**10.2.6.2. International Instruments**

There are a minimum of 16 international treaties and/or regulatory mechanisms that pertain to the East Indian-West Pacific DPS, and nearly all countries lining the East Indian and West Pacific Oceans have some level of national legislation directed at sea turtle protection. The international instruments listed below apply to sea turtles found in the Mediterranean Sea and are described in Appendix 5.

- Convention on Biological Diversity
- Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention)
- Convention on the Conservation of Migratory Species of Wild Animals
- Convention on International Trade in Endangered Species of Wild Fauna and Flora
- Convention for the Protection of the Natural Resources and Environment of the South Pacific Region
- Food and Agriculture Organization Technical Consultation on Sea Turtle-Fishery Interactions
10.3. Assessment of Significant Portion of its Range (SPR)

The SRT reviewed the information on threats and extinction risk within each DPS to determine whether there were portions of the range of the DPS that might be considered “significant portions of the range” in accordance with the definition of endangered and threatened species and the draft policy that interprets that term (see Section 3.4 for more details on the SPR deliberative process).

The East Indian-West Pacific DPS includes a large geographic area and total abundance is relatively large albeit much reduced from historical levels. Threats are fairly uniform throughout the region although conservation practices vary in implementation and effectiveness. Recent trends, which include nesting numbers recorded over the last 20–30 years, indicate that nesting females throughout the DPS are decreasing, with the exception of Sabah in Malaysia and Baguan in the Philippines, where the nesting trend is increasing, presumably due to effective conservation efforts. There are a few beaches in Vietnam, Taiwan, Malaysia, and the Philippines that show recent signs of stability. Western Australia has a high number of females although trends for this area are not known as sampling efforts were not consistent across the years.

The SRT concluded that, if the turtles in the rookeries that are currently known to be declining were lost, the remaining populations would be at greater risk of extinction. Therefore, the SRT concluded that the portion of the DPS with the declining populations may constitute a significant portion of its range. The next step in this analysis was to determine how extinction risk to the entire DPS would change if all these declining populations were lost.

10.4. Assessment of Extinction Risk

For the SRT’s assessment of extinction risk for green turtles in the East Indian-West Pacific DPS, there were two separate sets of ranking exercises: One focusing on the importance that each SRT member placed on each of the six different elements considered for this region (Table 10.3), and a second which reflects the SRT members’ expert opinion about the probability that green turtles would fall into any one of the various extinction probability ranges (Table 10.4;
see Section 3.3, Assessment of Extinction Risk Framework, for discussion of this process). Both of these exercises had to be completed twice, once for the entire DPS, and one for the DPS assuming the SPR was extirpated and only the nesting populations of currently stable or increasing beaches remained (see Section 3.4, Assessment of Significant Portion of its Range (SPR)).

10.4.1. Risk Assessment Voting For Entire DPS

The SRT first conducted voting on both the six elements and the overall risk of extinction for the entire DPS (Tables 10.3 and 10.4). See section 3.3. for details on the six elements and the voting process.

Table 10.3. Summary of ranks that reflect the importance placed by each SRT member on the critical assessment elements considered for the entire range of the East Indian-West Pacific DPS. For Elements 1–4, higher ranks indicate higher risk factors.

<table>
<thead>
<tr>
<th>Critical Assessment Elements</th>
<th>Element 1</th>
<th>Element 2</th>
<th>Element 3</th>
<th>Element 4</th>
<th>Element 5</th>
<th>Element 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abundance (1 to 5)</td>
<td>1.50</td>
<td>2.75</td>
<td>1.42</td>
<td>1.33</td>
<td>–1.42</td>
<td>0.50</td>
</tr>
<tr>
<td>Trends / Productivity (1 to 5)</td>
<td>0.19</td>
<td>0.22</td>
<td>0.19</td>
<td>0.19</td>
<td>0.23</td>
<td>0.19</td>
</tr>
<tr>
<td>Spatial Structure (1 to 5)</td>
<td>1–3</td>
<td>1–4</td>
<td>1–2</td>
<td>1–2</td>
<td>(-2)–0</td>
<td>0–2</td>
</tr>
<tr>
<td>Diversity / Resilience (1 to 5)</td>
<td>0.19</td>
<td>1.42</td>
<td>1.33</td>
<td>–1.42</td>
<td>0.50</td>
<td>0.19</td>
</tr>
<tr>
<td>Five-Factor Analyses (–2 to 0)</td>
<td>0.23</td>
<td>0.19</td>
<td>1–2</td>
<td>1–2</td>
<td>0.50</td>
<td>0.19</td>
</tr>
<tr>
<td>Conservation Efforts (0 to 2)</td>
<td>0.19</td>
<td>0.19</td>
<td>0.23</td>
<td>0.23</td>
<td>0.50</td>
<td>0.19</td>
</tr>
<tr>
<td>MEAN RANK</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>SEM</td>
<td>0.19</td>
<td>0.22</td>
<td>0.19</td>
<td>0.19</td>
<td>0.23</td>
<td>0.19</td>
</tr>
<tr>
<td>RANGE</td>
<td>1–3</td>
<td>1–4</td>
<td>1–2</td>
<td>1–2</td>
<td>(-2)–0</td>
<td>0–2</td>
</tr>
</tbody>
</table>

With respect to the important rankings for the six critical assessment elements, nesting trends (Element 2) and the 5-Factor Analysis (threats) featured most prominently in the risk threshold voting. Spatial structure (i.e., widespread overall nesting distribution) and diversity/resilience (i.e., high genetic diversity) featured relatively low in the risk threshold voting, likely resulting from the geographically widespread nature of the DPS, along with substantial nesting beaches occurring across the DPS. SRT members also generally thought that future threats not yet reflected in nester abundance or not yet experienced by the population weighed heavier in their risk assessment voting than did any conservation efforts that are not yet reflected in nester abundance. With respect to the diversity of opinions among the SRT members when considering the six Critical Elements, a large range in rankings (i.e., voter opinion) was noted for all the 5-Factor Analysis and Conservation Efforts (w/ ranks from 0 to –2 and 0 to 2 respectively).
Table 10.4. Summary of Green Turtle SRT member expert opinion about the probability that the East Indian-West Pacific DPS will reach a critical risk threshold within 100 years. Each SRT member assigned 100 points across the rank categories. The continuum in Row 1 has categories with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability Of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1-5%</th>
<th>6-10%</th>
<th>11-20%</th>
<th>21-50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN ASSIGNED POINTS</td>
<td>60.50</td>
<td>12.75</td>
<td>10.25</td>
<td>9.83</td>
<td>4.17</td>
<td>2.50</td>
</tr>
<tr>
<td>SEM</td>
<td>10.97</td>
<td>3.64</td>
<td>3.68</td>
<td>4.90</td>
<td>2.94</td>
<td>2.09</td>
</tr>
<tr>
<td>Min</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Max</td>
<td>99</td>
<td>30</td>
<td>40</td>
<td>50</td>
<td>35</td>
<td>25</td>
</tr>
</tbody>
</table>

With respect to the overall risk of extinction, of the critical risk threshold categories describing the probability that the DPS will reach a critical risk threshold within 100 years (Table 10.4), the SRT member votes resulted in the greatest point (i.e. probability) designations in the '<1%' and '1–5%' risk ranges (mean of 60.50 and 12.75 points, respectively). The '6–10%' and '11–20%' received 10.25 points and 9.83 points respectively while the '21–50%' and '>50%' ranges received the fewest points from SRT members (mean of 4.17 and 2.5 respectively).

In their vote justifications, most members cited the widespread nesting area throughout the DPS, high abundance, and high level of genetic diversity and resilience. There was concern about trends/productivity with many of the higher abundance rookeries exhibiting decreasing trends, though there appears to be stable or increasing trends at five nesting sites. SRT members also cited high levels of threats, which include heavy poaching and illegal trafficking in some areas of the DPS and continued harvesting of turtles and eggs, cumulative fisheries bycatch without adequate conservation measures in place.

10.4.2. Extinction Risk with SPR consideration

Because the SRT determined that an SPR potentially exists within this DPS, the SRT also had to repeat the voting on both the six elements and the overall risk of extinction, assuming that the SPR (declining populations) was lost. See section 3.3. for details on the six elements and the voting Terms of Reference.

Table 10.5. Summary of ranks that reflect the importance placed by each SRT member on the critical assessment elements considered when voting on extinction risk for the East Indian-West Pacific green turtle DPS, assuming the SPR is lost. For Elements 1–4, higher ranks indicate higher risk factors.
For the SRT’s assessment of extinction risk with SPR considerations (i.e., SPR extirpated), concerns about abundance (Element 1), spatial structure (Element 3), and diversity/resilience (Element 4) increased somewhat from considerations without the SPR extirpated. This naturally follows a loss in populations that are range throughout the DPS. Concerns about trends (Element 2) and the 5-factors (or threats not yet reflected in nester abundance or not yet experienced by the population) decreased somewhat. This is likely due to loss of the only decreasing populations, which are likely those facing the greatest threats. The diversity of expert opinion, as reflected in the range in rankings, increased for Elements 1, 3, and 4 (indicating that concern increased for some members and not others), and remained the same for Element 2 (Trends) and for the at 0 to –2 and 1 to 2 for the 5-Factor Analysis and Conservation Efforts, all of which already had a high range (1 to 4, 0 to –2, and 0 to 2, respectively).

Table 10.6. Summary of Green Turtle SRT member expert opinion about the probability that the East Indian-West Pacific DPS will reach a critical risk threshold within 100 years, without the SPR. Each SRT member assigned 100 points across the rank categories. The continuum in Row 1 has categories with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1-5%</th>
<th>6-10%</th>
<th>11-20%</th>
<th>21-50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN ASSIGNED POINTS</td>
<td>52.00</td>
<td>13.73</td>
<td>13.73</td>
<td>9.64</td>
<td>7.27</td>
<td>3.64</td>
</tr>
<tr>
<td>SEM</td>
<td>11.31</td>
<td>3.17</td>
<td>5.57</td>
<td>3.60</td>
<td>4.18</td>
<td>3.17</td>
</tr>
<tr>
<td>Min</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Max</td>
<td>95</td>
<td>30</td>
<td>60</td>
<td>30</td>
<td>45</td>
<td>35</td>
</tr>
</tbody>
</table>

With respect to the risk of extinction with SPR consideration, of the critical risk threshold categories describing the probability that the DPS will reach a critical risk threshold within 100 years (Table 10.6), the SRT member votes resulted in the greatest point (i.e., probability) designations in the ‘<1%’ and ‘1–5%’ risk ranges (mean of 52 and 13.73 points, respectively), although the <1 percent category decreased substantially.

The combined expert judgment of the SRT is that the DPS would be at an increased risk of extinction if the SPR was lost, with 52 vs. 60.5 percent chance that the population has a ‘<1%’
risk of extinction, and a 48 vs. 39.5 percent chance that the population has ‘>1%’ risk of extinction. This appears to be due to increases in risk in the elements of Abundance, Spatial Structure and Diversity / Resilience.

In their vote justifications, members cited the large number of females present at various locations throughout the DPS, weighed against the continuing harvest of eggs and turtles and fisheries bycatch on the remaining portions of DPS and the substantial loss of diversity and connectivity. The unknown trend in Australia was also considered.

10.5. Synthesis and Integration

The East Indian-West Pacific DPS is characterized by a relatively large geographic area with widespread nesting reported in 58 different locations throughout the DPS. Although the SRT rated the abundance element of relatively low concern in its ranking of the Critical Assessment elements (1.5 out of 5), decades of harvesting and habitat degradation have led to a drastic decline in the sea turtle populations within this DPS in the last century. Population trends at many of the higher abundance rookeries are decreasing, though there appears to be an increasing trend on Sabah in Malaysia and on Baguan in the Philippines, presumably due to effective conservation efforts. As such, the trends/productivity element ranked as a higher risk (2.75 out of 5). Spatial structure and diversity/resilience in the East Indian-West Pacific DPS were considered by the SRT to have a relatively low likelihood of contributing to the extinction of the DPS in the next 100 years (1.6 and 1.5 out of 5, respectively).

Continued harvest, coastal development, beachfront lighting, erosion, fishing practices, and marine pollution both at nesting beaches and important foraging grounds are all continuing concerns across the DPS. Harvest of turtles and eggs for human consumption continues as a high threat to this East Indian-West Pacific DPS. Coastal development, largely due to tourism, is an increasing threat in many areas. Fishery bycatch occurs throughout the DPS, particularly bycatch mortality of green turtles from pelagic longline, set net, and trawl fisheries. Additional threats due to climate change, such as loss of habitat due to sea level rise and increased production of female turtles, negatively impact this DPS. Conservation efforts have been effective in a few areas but are lacking or not effective in most.

The SRT considered all of the above in the overall critical risk threshold. Approximately 16.5 percent of the votes cast for were for the >11 percent likelihood of reaching a critical risk threshold of extinction within 100 years, with 23 percent cast for 1–10 percent likelihood, and 60.5 percent cast for <1 percent likelihood of reaching a critical risk threshold. These results reflect the widespread nesting area throughout the DPS, relatively high remaining abundance of turtles, and high level of genetic diversity, but also concern about trends/productivity, with many of the higher abundance nesting sites exhibiting decreasing trends, and substantial ongoing threats to the DPS.
11. CENTRAL WEST PACIFIC DPS (DPS #7)

11.1. DPS Range and Nesting Distribution

The Central West Pacific DPS encompasses the Republic of Palau (Palau), Federated States of Micronesia, New Guinea, Solomon Islands, Marshall Islands, Guam, the Commonwealth of the Northern Mariana Islands (CNMI), and a portion of Japan (Ogasawara; Figure 11.1).

![Map of nesting distribution](image)

**Figure 11.1.** Nesting distribution of green turtles in the Central West Pacific DPS (blue-shaded region). Size of circles indicates nesting abundance category. Locations marked with ‘×’ indicate nesting sites lacking abundance information.

Green turtle nesting occurs at least at low levels throughout the geographic distribution of the population, with isolated locations having high nesting activity. The highest numbers of females nesting in this DPS are located in Gielop and Iar Island, Ulithi Atoll, Yap, Federated States Of Micronesia (1,412); Bikar Atoll, Marshall Islands (300); Merir Island, Palau (441); and Chichijima (1,301) and Hahajima (394), Ogasawara, Japan (Barr, 2006; Palau BMR, 2008, 2005; Maison et al., 2010; NMFS and FWS, 1998; H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012; J. Cruce, Ocean Society, pers. comm., 2013).

There are numerous other populations in the Federated States of Micronesia, Solomon Islands, and Palau, and approximately 22 nesting female green turtles in Guam, and 57 nesting female green turtles in CNMI. Historical baseline nesting information in general is not widely available.
in this region, but exploitation and trade of green turtles throughout the region is well-known (Groombridge and Luxmoore, 1989).

Green turtles departing nesting grounds in this DPS travel throughout the western Pacific Ocean. Results of three post-nesting green turtles from Palau in 2006 showed they remained nearby or traveled to the Aru Islands in Indonesia – roughly 1,100 km away (Klain et al., 2007). Five post-nesting green turtles leaving Erikub Atoll in the Marshall Islands in 2007 traveled to the Philippines, Kiribati, Federated States Of Micronesia, or remained in the Marshallese EEZ (Kabua et al., 2012). Turtles tagged in Yap (Federated States of Micronesia) were recaptured in the Philippines, Marshall Islands, Papua New Guinea, Palau, and Yap (Palau BMR, 2008; Cruce, 2009). A turtle tagged on Gielop Island, Yap in 1991 was recaptured in Muroto Kochi prefecture, Japan in 1999 (Miyawaki et al., 2000). A nesting female tagged on Merir Island, Palau was captured near the village of Yomitan Okinawa, Japan (Palau BMR, 2008). Hundreds of nesting females tagged in Ogasawara Island were recaptured in the main islands of Japan, the Ryukyu Archipelago (Okinawa), Taiwan, China, and Philippines (Suganuma personal communication; Ogasawara Marine Station, Everlasting Nature of Asia unpublished data). A turtle tagged in Japan was recorded nesting in Yap, Federated States of Micronesia (Cruce, 2009).

In addition to nesting beaches, green turtles are found in coastal waters in low to moderate densities at foraging areas throughout the DPS. Aerial sea turtle surveys show that an in-water population exists around Guam (DAWR, 2011). In-water green turtle density in the Marianas Archipelago is low and mostly restricted to juveniles (Pultz et al., 1999; Kolinski et al., 2006, 2005; Palacios, 2012a). In-water information in this DPS overall is particularly limited.

11.2. Critical Assessment Elements

In the evaluation of extinction risk for green turtles in the Central West Pacific DPS, the SRT considered six distinct elements, each of which are discussed below: (1) Nesting Abundance, (2) Population Trends, (3) Spatial Structure, (4) Diversity / Resilience, (5) Five-Factor Threat Analyses, and (6) Existing Conservation Efforts. See Section 3.3 for additional information on the selection of these six elements.

11.2.1. Nesting Abundance

Currently, there are approximately 51 nesting sites and 6,518 nesting females in the Central West Pacific (Table 11.1 and Table 11.2). There are a number of unquantified nesting sites, possibly with small numbers, however specifics regarding these sites is unknown. The largest nesting site is in the Federated States of Micronesia, and that particular site hosts approximately 22 percent of the total annual nesting females for this DPS.


RESPONSE: Added. Searching pdf to send to Camryn.

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Comment [A2]: Yomitan….

RESPONSE: Changed.

Comment [A3]: Hundreds of nesting females tagged in Ogasawara Island were recaptured in the main islands of Japan, the Ryukyu Archipelago (Okinawa), Taiwan, China, and Philippines (Suganuma personal communication; Ogasawara Marine Station, Everlasting Nature of Asia unpublished data)

RESPONSE: Added.
Table 11.1. Summary of green turtle nesting sites in the Central West Pacific DPS. Data are organized by country, nesting site, monitoring period, and estimated total nester abundance [(Total Counted Females/Years of Monitoring) x Remigration Interval]. For a list of references for these data, see Appendix 2.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YEARS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Federated States of Micronesia</td>
<td>Fanang</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>Federated States of Micronesia</td>
<td>Gaferut</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>Federated States of Micronesia</td>
<td>Orolok Atoll</td>
<td>1990</td>
<td>n/a</td>
</tr>
<tr>
<td>Federated States of Micronesia</td>
<td>Pikelic Atoll</td>
<td>1970</td>
<td>n/a</td>
</tr>
<tr>
<td>Federated States of Micronesia</td>
<td>Sorol Atoll</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>Marshall Islands</td>
<td>Ailuk</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>Marshall Islands</td>
<td>Wotho</td>
<td>1988</td>
<td>n/a</td>
</tr>
<tr>
<td>Marshall Islands</td>
<td>Wotje Atoll</td>
<td>2003</td>
<td>n/a</td>
</tr>
<tr>
<td>Papua New Guinea</td>
<td>Atmago (Egmakau)</td>
<td>2007</td>
<td>n/a</td>
</tr>
<tr>
<td>Papua New Guinea</td>
<td>Emirau</td>
<td>2007</td>
<td>n/a</td>
</tr>
<tr>
<td>Papua New Guinea</td>
<td>Lemus</td>
<td>2007</td>
<td>n/a</td>
</tr>
<tr>
<td>Papua New Guinea</td>
<td>Mussau</td>
<td>2007</td>
<td>n/a</td>
</tr>
<tr>
<td>Papua New Guinea</td>
<td>Nago</td>
<td>2007</td>
<td>n/a</td>
</tr>
<tr>
<td>Papua New Guinea</td>
<td>Nusalaman (Nusalomon)</td>
<td>2007</td>
<td>n/a</td>
</tr>
<tr>
<td>Papua New Guinea</td>
<td>Ral</td>
<td>2007</td>
<td>n/a</td>
</tr>
<tr>
<td>Papua New Guinea</td>
<td>Usen (Usang)</td>
<td>2007</td>
<td>n/a</td>
</tr>
<tr>
<td>Palau</td>
<td>Pulo Ana Island</td>
<td>2005</td>
<td>1</td>
</tr>
<tr>
<td>Palau</td>
<td>Kayangel Atoll</td>
<td>2005</td>
<td>6</td>
</tr>
<tr>
<td>Palau</td>
<td>Ngarchelung State</td>
<td>2005</td>
<td>6</td>
</tr>
<tr>
<td>Palau</td>
<td>Ngerechur Island</td>
<td>2005</td>
<td>6</td>
</tr>
<tr>
<td>Federated States of Micronesia</td>
<td>Murilo Atoll</td>
<td>1993</td>
<td>9</td>
</tr>
<tr>
<td>Indonesia</td>
<td>Jamursba-Medi</td>
<td>1995-1997</td>
<td>9</td>
</tr>
<tr>
<td>Solomon Islands</td>
<td>Hakelake Island</td>
<td>1995</td>
<td>11</td>
</tr>
<tr>
<td>CNMI</td>
<td>Rota</td>
<td>2012</td>
<td>15</td>
</tr>
<tr>
<td>CNMI</td>
<td>Tinian</td>
<td>2012</td>
<td>15</td>
</tr>
<tr>
<td>Federated States of</td>
<td>East Fayu</td>
<td>1993</td>
<td>18</td>
</tr>
<tr>
<td>COUNTRY</td>
<td>NESTING SITE</td>
<td>MONITORING PERIOD (YEARS)</td>
<td>ESTIMATED NESTER ABUNDANCE</td>
</tr>
<tr>
<td>--------------------------</td>
<td>-------------------------------------</td>
<td>---------------------------</td>
<td>---------------------------</td>
</tr>
<tr>
<td>Micronesia</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Marshall Islands</td>
<td>Rongerik Atoll</td>
<td>2003</td>
<td>21</td>
</tr>
<tr>
<td>Guam</td>
<td>Island of Guam (and Cocos)</td>
<td>2010; 2008-2010</td>
<td>22</td>
</tr>
<tr>
<td>CNMI</td>
<td>Saipan</td>
<td>2012</td>
<td>27</td>
</tr>
<tr>
<td>Solomon Islands</td>
<td>Kerehikapa Island</td>
<td>1995</td>
<td>32</td>
</tr>
<tr>
<td>Japan</td>
<td>Mukojima</td>
<td>2010-2012</td>
<td>39</td>
</tr>
<tr>
<td>Marshall Islands</td>
<td>Bikini</td>
<td>1992</td>
<td>75</td>
</tr>
<tr>
<td>Marshall Islands</td>
<td>Enewetak</td>
<td>1992</td>
<td>75</td>
</tr>
<tr>
<td>Marshall Islands</td>
<td>Erikub</td>
<td>1992</td>
<td>75</td>
</tr>
<tr>
<td>Marshall Islands</td>
<td>Jemo</td>
<td>1992</td>
<td>75</td>
</tr>
<tr>
<td>Federated States of Micronesia</td>
<td>Olimarao Atoll</td>
<td>1990</td>
<td>81</td>
</tr>
<tr>
<td>Federated States of Micronesia</td>
<td>Elato Atoll</td>
<td>1993</td>
<td>90</td>
</tr>
<tr>
<td>Palau</td>
<td>Helen Island</td>
<td>2005</td>
<td>141</td>
</tr>
<tr>
<td>Federated States of Micronesia</td>
<td>Ngulu Atoll</td>
<td>1993</td>
<td>192</td>
</tr>
<tr>
<td>Solomon Islands</td>
<td>Ausilala</td>
<td>1981</td>
<td>225</td>
</tr>
<tr>
<td>Solomon Islands</td>
<td>Balaka</td>
<td>1981</td>
<td>225</td>
</tr>
<tr>
<td>Solomon Islands</td>
<td>Maifu</td>
<td>1981</td>
<td>225</td>
</tr>
<tr>
<td>Solomon Islands</td>
<td>Malaulaul</td>
<td>1981</td>
<td>225</td>
</tr>
<tr>
<td>Solomon Islands</td>
<td>Malaupaina</td>
<td>1981</td>
<td>225</td>
</tr>
<tr>
<td>Solomon Islands</td>
<td>Wagina</td>
<td>1981</td>
<td>225</td>
</tr>
<tr>
<td>Federated States of Micronesia</td>
<td>Ulithi Atoll</td>
<td>2010-2012</td>
<td>280</td>
</tr>
<tr>
<td>Marshall Islands</td>
<td>Bikar Atoll</td>
<td>1992</td>
<td>300</td>
</tr>
<tr>
<td>Japan</td>
<td>Hahajima</td>
<td>2010-2012</td>
<td>394</td>
</tr>
<tr>
<td>Palau</td>
<td>Merir Island, Sonsorol State</td>
<td>November 2007 to August 2008</td>
<td>441</td>
</tr>
<tr>
<td>Japan</td>
<td>Chichijima</td>
<td>2010-2012</td>
<td>1,301</td>
</tr>
<tr>
<td>Federated States of Micronesia</td>
<td>Ulithi Atoll</td>
<td>2010-2012</td>
<td>1,412</td>
</tr>
<tr>
<td></td>
<td>Gielop and Iar Island</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>
Table 11.2. Green turtle nester abundance distribution among nesting sites in the Central West Pacific.

<table>
<thead>
<tr>
<th>NESTER ABUNDANCE</th>
<th># NESTING SITES</th>
</tr>
</thead>
<tbody>
<tr>
<td>n/a</td>
<td>16</td>
</tr>
<tr>
<td>1 to 10</td>
<td>6</td>
</tr>
<tr>
<td>11-50</td>
<td>9</td>
</tr>
<tr>
<td>51-100</td>
<td>6</td>
</tr>
<tr>
<td>101-500</td>
<td>12</td>
</tr>
<tr>
<td>501-1000</td>
<td>0</td>
</tr>
<tr>
<td>1001-5000</td>
<td>2</td>
</tr>
<tr>
<td>5001-10000</td>
<td>0</td>
</tr>
<tr>
<td>&gt;10000</td>
<td>0</td>
</tr>
<tr>
<td><strong>TOTAL SITES</strong></td>
<td><strong>51</strong></td>
</tr>
<tr>
<td><strong>TOTAL ABUNDANCE</strong></td>
<td><strong>6,518</strong></td>
</tr>
<tr>
<td><strong>PERCENTAGE at LARGEST NESTING SITE</strong></td>
<td><strong>21.66% (Federated States of Micronesia)</strong></td>
</tr>
</tbody>
</table>

11.2.2. Population Trends

There is insufficient long-term and standardized monitoring information to adequately describe abundance and population trends for many areas of the Central West Pacific DPS. The limited available information suggests a nesting population decrease in some portions of the DPS like the Marshall Islands, or unknown trends in other areas such as Palau, Papua New Guinea, the Marianas, Solomon Islands, or the Federated States of Micronesia (Maison et al., 2010). For a list of references on trend data, see Appendix 3.

PVAs were conducted for nesting sites that had a minimum of 15 years of recent nesting data with an annual nesting level of more than 10 females (for more on data quantity and quality standards used, see Section 3.2). To assist in interpreting these PVAs, we indicate the probability of green turtle nesting populations declining to two different biological reference points, one using a trend-based and the other an abundance-based threshold. The trend-based reference point for evaluating population forecasts is half of the last observed abundance value, i.e., a 50 percent decline. The abundance-based reference point was a total adult female abundance of 300 females. Risk is calculated as the percentage of model runs that fall below these reference points after 100 years. For a full discussion of these PVAs and these reference points, see Section 3.2.

There is only one site for which 15 or more years of recent data are available for annual nester abundance (one of the standards for representing PVAs in this report). This is at Chichijima, Japan, one of the major green turtle nesting concentrations in Japan (Horikoshi et al., 1994;
This PVA has limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. The PVA suggests a zero percent probability that this population will fall below the trend reference or absolute abundance reference in 100 years. The population has increased from a mean of approximately 100 females/year in the late 1970s/early 1980s to a mean of approximately 500 per year since 2000. Chaloupka et al. (2008) reports an estimated annual population growth rate of 6.8% per year for the Chichijima nesting site.

**Figure 11.2.** Stochastic Exponential Growth (SEG) Model Output for Chichijima, Japan. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference. Nesters were computed from nest counts using 4.1 nests per female (Suganuma et al., 1996).

### 11.2.3. Spatial Structure

When examining spatial structure for the Central West Pacific DPS, the SRT examined genetic data, flipper and satellite tagging, and demographic data. Genetic sampling in the Central West Pacific has recently improved, but remains challenging given the large number of small island and atoll nesting sites. Stock structure analysis indicated that rookeries separated by more than 1000 km were significantly differentiated from each other (FST values from 0.06 – 0.9, p<0.001) while neighboring rookeries within 500 km showed no genetic differentiation. Dutton et al. (unpublished data) suggest that there are at least seven independent stocks in the region.

With respect to flipper tagging, there are records of turtles tagged in the Philippines nesting in the Federated States of Micronesia; a turtle tagged in Japan was recorded nesting in the Federated States of Micronesia; turtles tagged in the Japan Archipelago and China were recorded nesting in the Ogasawara islands (Suganuma personal communication; Ogasawara Marine Center, Everlasting Nature of Asia unpublished data); and turtles tagged...
Satellite telemetry shows that nesting females migrate to areas within and outside of the Central
West Pacific DPS. For example, satellite tracks show animals moving from the Mariana Islands
to the Philippines and Japan, and others moving from the Chichijima Islands of Ogasawara to the
main islands of Japan (Japan Fisheries Resource Conservation Association, 1999; Hatase et al.,
2006). Green turtles have also been shown to move from the Federated States of Micronesia to
the Philippines and to the west (G. Balazs, NMFS, unpublished data; Kolinski, et al. unpublished
data.)

Demographic data availability is limited and variable for many nesting sites in this DPS. Overall
the demographics of green turtles in the Central West Pacific vary among the different nesting
assemblages, for a variety of demographic parameters. This variability in parameters such as
remigration interval, clutch size, hatching success, sex ratio and clutch frequency is not separated
out regionally within the DPS and, therefore, does not necessarily suggest a high level of
population structuring. Hatching success varies widely from 44.1-73.8 percent for areas with
available information (Suganuma, 1985). Clutch size range varies widely from 59 to 139
eggs/nest (Palacios, 2012a, 2012b). Clutch frequency ranges from 4 to 6 nests per season
(Suganuma, 1985). Remigration interval varies from 3 to 4 years by nesting site (Cruce, 2009).
The nester sizes range from 102cm CCL in Palau to 103.4cm CCL in Yap, Federated States of
Micronesia Palau (Palau BMR, 2008; Cruce, 2009))

11.2.4. Diversity / Resilience

The aspects considered under this critical element include the overall nesting spatial range,
diversity in nesting season, diversity of nesting site structure and orientation (e.g., insular vs.
continental nesting sites), and the genetic diversity within the DPS. Aspects such as these are
important considerations for assessing the potential impact of catastrophic events such as storms,
sea level rise, and disease.

The overall range of the DPS is relatively widespread. Green turtles in this DPS are found from
the Marshall Islands in the east to Palau in the west, and from Japan in the north to the Solomon
Islands in the south. Nesting occurs on various islands and atolls throughout the DPS, however
at what appears to be low numbers (except for a few locations). Nesting information is limited
for some areas, however occurs from November to August in Palau; from March through
September in the Federated States of Micronesia; and May to August in Ogasawara, Japan.
Some animals are traveling outside the bounds of this DPS, into the East Indian/West Pacific
DPS.

While nesting and foraging areas are not concentrated in one area and this provides a level of
habitat use diversity and population resilience, the contribution of this characteristic to such
diversity and resilience is reduced by the threats faced in each of the nesting and foraging areas.

11.2.5. Analysis of Factors Listed Under ESA Section 4(a)(1)
Section 4(a)(1) of the ESA lists five factors (threats) that must be considered when determining the status of a species. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

The following information on these factors/threats pertains to green turtles found in the Central West Pacific DPS.

11.2.5.1. **Factor A: Destruction or Modification of Habitat or Range**

The Central West Pacific DPS of the green turtle is negatively affected by ongoing changes in both its terrestrial and marine habitats as a result of land and water use practices as considered above in Factor A. Within Factor A, we find that coastal development, beachfront lighting, and erosion resulting from sand mining, and fishing practices, marine pollution, and climate change continue as threats to this DPS.

**Terrestrial Zone**

In the Central West Pacific Ocean, some nesting beaches have become severely degraded from a variety of activities. Destruction and modification of green turtle nesting habitat result from coastal development and construction, placement of barriers to nesting, beachfront lighting, vehicular and pedestrian traffic, sand extraction, beach erosion, beach pollution, removal of native vegetation, and presence of non-native vegetation.

Human populations are growing rapidly in many areas of the insular Pacific and this expansion is exerting increased pressure on limited island resources. The most valuable land on most Pacific islands is often located along the coastline, particularly when it is associated with a sandy beach. Construction is occurring at a rapid rate in some areas and is resulting in loss or degradation of green turtle nesting habitat. Construction-related threats to the region’s green turtle nesting beaches include the construction of buildings (e.g., hotels, houses, restaurants) and recreational facilities (e.g., golf courses) on or directly adjacent to the beach; clearing stabilizing beach vegetation (which accelerates erosion); and the use of heavy construction equipment on the beach, which can cause sand compaction or beach erosion. Lighting associated with coastal development is also degrading nesting habitat. Security and street lights, restaurant, hotel and other commercial lights, and recreational lights misdirect hatchlings throughout the Central West Pacific every year. Additional threats to green turtle nesting habitat include increased recreational and commercial use of beaches, the loss of nesting habitat to human activities (e.g., pig pens on beaches), beach camping and fires, and an increase in litter and other refuse. Weather events, such as storms, and seasonal changes in current patterns can also reduce or eliminate sandy beaches, degrade turtle nesting habitat, and cause barriers to adult and hatchling turtle movements on affected beaches.
On Saipan, Tinian, and Rota Islands in the CNMI, coastal development and ensuing human activities impact green turtle nesting beach habitat (NMFS and USFWS, 1998). On Saipan, golf course, hotel, and tourism-related development has severely impacted most of the historical nesting areas on the western portion of the island, and residential development is threatening the eastern portion of the island. On Tinian, the majority of the nesting beaches are on military-leased land where the potential for construction impacts exist (CNMI Coastal Resources Management Office, 2011). Expected military expansion plans for the region are likely to include relocation of thousands of military personnel to Guam and increased training exercises in the CNMI (CNMI Coastal Resources Management Office, 2011). The U.S. military has identified areas on both Tinian and Pagan Islands where significantly increased training exercises would occur. The extent to which this proposed military expansion will affect sea turtle nesting habitat is uncertain. On Rota, green turtle nesting appeared to be limited to undeveloped private land due to heavy recreational use and tourist developments on remaining beaches; however, many of the undeveloped beaches were believed likely to be eventually developed (NMFS and USFWS, 1998).

In the Federated States of Micronesia, construction of houses and pig pens on Oroluk beaches in Pohnpei State interferes with turtle nesting by creating barriers to nesting habitat (NMFS and USFWS, 1998; Buden, 1999). Nesting habitat destruction is also a major threat to Guam turtles and has resulted mainly from construction and development due increased tourism (NMFS and USFWS, 1998; Project GloBAL, 2009a). Coastal construction is a moderate problem on Majuro Atoll in the Republic of the Marshall Islands (NMFS and USFWS, 1998); however, it is unknown to what extent nesting beaches are being affected. On the outer atolls of the Marshall Islands, beach erosion has been aggravated by airfield and dock development, and by urban development on Majuro and Kwajalein Atolls. In the Republic of Palau, increasing nesting habitat degradation from tourism and coastal development has been identified as a threat to sea turtles (Isamu and Guilbeaux, 2002; Eberdong and Klain, 2008), although the extent and significance of the impacts are unknown.

As indicated above, coastal development is usually accompanied by artificial lighting. In the CNMI, beachfront lighting was identified in 1998 as a high potential future problem in Rota where resort development was flourishing (NMFS and USFWS, 1998); however, information is not available to determine if this is now a problem on Rota. Most houses and hotels adjacent to the lagoon area of Saipan usually have some form of beach lighting. In 2011, CNMI Division of Fish and Wildlife staff identified lighting problems, including resort and housing development lighting, beach bonfires, campers with lanterns, and shore fishermen with flashlights, on five nesting beaches (Wing Beach, Lao Lao Bay, Tank Beach, Coral Ocean Point Beach, and Bird Island Beach) on Saipan (Palacios, 2012a). In addition, cumulative lighting from resort and housing developments has created a sky glow affect near some nesting beaches. However, as of the 2011 nesting season, no nesting or hatchling turtle lighting disorientations had been documented on Saipan.

In 1994, the village of Melekeok (Melekeok State) in Palau reported that green turtle hatchlings were attracted into lighted houses and to street lights (M. Guilbeaux pers. comm. cited in NMFS and USFWS 1998). In addition, campfires and houselights are a problem at Angaur, Peleliu, Kayangel and the Southwest Island beaches on Guam (NMFS and USFWS, 1998). Beachfront
lighting was not harmful in the Federated States of Micronesia in 1998 (NMFS and USFWS, 1998); however, more recent information is unavailable to determine if this is still the case. Lighting on Guam is a problem with unknown impact (NMFS and USFWS, 1998), although Navy (2005 cited in Project GloBAL 2009a) states that sea turtle nesting beaches in Guam are impacted by the presence of high intensity lighting. In the Republic of the Marshall Islands, portions of Majuro and Kwajalein are lighted, but the impact is unknown (NMFS and USFWS, 1998).

Beach mining occurs on the island of Falalop within the Ulithi Atoll in Yap State and in Pohnpei and Chuuk States and may occur on other inhabited islands, but did not appear to be a major problem in the 1990s (NMFS and USFWS, 1998). However, Smith et al. (1997) indicated that beach sands are a favored source of sand for use as construction aggregate, but that in the absence of beaches, sand is extracted from reef-derived sand in the lagoon surrounding Pohnpei Island instead. In the Republic of the Marshall Islands, mining of beach sand has been identified as a serious problem on Majuro Atoll (NMFS and USFWS, 1998; Hay and Sablan-Zebedy, 2005), although in 1998 it was reported as not being a problem at the only known green turtle nesting beach (Iroij Island) (NMFS and USFWS, 1998). Mining of beach sand for use in construction has been identified as a threat to beaches in the Gilbert Islands chain in Kiribati (Kiribati Ministry of Environment, Lands and Agriculture Development, 2009), and has the potential to impact nesting and hatching green turtles and nests. Beach sand mining, coupled with increases in storm surge intensity and removal of coastal vegetation is causing accelerated erosion of coastlines. However, in 2012, it was announced that beach sand mining on South Tarawa, Kiribati’s most populated atoll island, would soon be phased out and replaced by lagoon dredging due to the severe coastal erosion problems caused by beach sand mining (Pacific News Center, 2012).

Increased public use of nesting beaches is a threat to sea turtle nesting habitat in the CNMI. Public use of beaches includes a variety of recreational activities, including picnicking, swimming, surfing, playing sports, scuba diving and snorkeling access (CNMI Coastal Resources Management Office 2011). Also in the CNMI, beach driving is a pastime on Saipan (NMFS and USFWS, 1998; Palacios, 2012a); however, the impact of this activity on green turtle nesting habitat is unknown. Although CNMI public law No. 11-61 prohibits motor vehicles from driving on any beach area, public driving on the beach still occurs (CNMI Coastal Resources Management Office 2011). Although driving on the Guam’s beaches is illegal, there is extensive vehicle traffic that is likely degrading green turtle nesting habitat (NMFS and USFWS, 1998; Wusstig, 2012).

In the CNMI, non-native vegetation, such as tangan tangan (Leucaena leucocephala) and devil’s gut (Cassytha filiformis), has been documented as creating an impediment to nesting turtles on beaches in Saipan (Palacios, 2012a). Also in the CNMI, marine debris was determined to be a pervasive issue at Tank Beach, Bird Island Beach, and Old Man by the Sea Beach on Saipan, although organized beach clean-ups have been conducted to help mitigate this impact (Palacios, 2012a).
Neritic/Oceanic Zones

Threats to habitat in the green turtle neritic and/or oceanic zones include fishing practices, channel dredging, sand extraction, marine pollution, and climate change. These threats also occur in the Central West Pacific Ocean.

Fishing methods not only incidentally capture green turtles, but also deplete invertebrate and fish populations and thus alter ecosystem dynamics. In many cases green turtle foraging areas coincide with fishing zones. However, comprehensive data currently are unavailable to fully understand how intense harvesting of fish resources changes neritic and oceanic ecosystems. Dynamite fishing occurs in the Federated States of Micronesia (NMFS and USFWS, 1998; Government of the Federated States of Micronesia, 2004) and the Republic of the Marshall Islands (Hay and Sablan-Zebedy, 2005). Dynamite fishing, as well as use of fish poisons, occurs in Papua New Guinea, although these practices are small scale and relatively isolated (Berdach and Mandeakali, 2004). These destructive fishing methods affect neritic zones by not only destroying bottom habitat, including seagrasses, but also by depleting fish populations and thus altering ecosystem dynamics.

In the CNMI, seagrass beds used by green turtles as foraging habitat have been identified on Saipan (Kolinski et al., 2001), Tinian (Kolinski et al., 2004), and Rota (Kolinski et al., 2006) Islands. Seagrasses around Tinian and Rota Islands have been reported as being in good condition, while seagrasses around Saipan have been reported as being degraded by hotels, golf courses, and general tourist activities (Project GloBAL, 2009b), presumably as a result of runoff and other impacts. Coastal development in Guam has resulted in sedimentation, which has damaged Guam’s coral reefs and, presumably, food sources for turtles (NMFS and USFWS, 1998).

Coral reefs and seagrass beds have also been severely degraded within the urban centers of the four states of the Federated States of Micronesia: Pohnpei, Yap, Chuuk, and Kosrae (NMFS and USFWS, 1998). Coral reefs and seagrass habitat off the lagoon shoreline of the Kwajalein Atoll islands and Majuro Atoll have been degraded by coastal construction, dredging, boat anchoring, and/or eutrophication from sewage and runoff from landfills, grave sites, and pig and chicken pens (NMFS and USFWS, 1998; Hay and Sablan-Zebedy, 2005).

Dredging and filling have contributed to changes to longshore processes and coastal erosion in the Marshall Islands (NMFS and USFWS, 1998; Hay and Sablan-Zebedy, 2005). A 1997 study found that most of the ocean and lagoon coastlines of Majuro Atoll are erosional with a shoreline retreat of 10 to 20 m having occurred in some places over an approximate 25-year period (Secretariat of the Pacific Community, 1997); this has implications for all of the Marshall Islands (Hay and Sablan-Zebedy, 2005). Dredging and sand mining has also been identified as a serious problem in Chuuk, Kosrae, and Yap States (Government of the Federated States of Micronesia,
For instance, on Pohnpei Island, sand is extracted from reef-derived sand in the lagoon surrounding the island resulting in the loss or degradation of seagrass meadows likely used as foraging habitat by green turtles (Smith et al., 1997).

In Kiribati’s Gilbert Islands chain, it was announced in 2012 that beach sand mining on South Tarawa, Kiribati’s most populated atoll island, would soon be phased out and replaced by lagoon dredging due to the severe coastal erosion problems caused by beach sand mining (Pacific News Center, 2012). While this is good news for the nesting beach habitat, lagoon dredging has the potential to negatively impact green turtle foraging habitat surrounding this island. Offshore sand mining also occurs in Palau, with possible implications for foraging habitat degradation (NMFS and USFWS, 1998). Green turtles used to be found foraging on seagrass beds at the mouth of Lighthouse Channel (a sand mining site on the eastern side of Malakal Harbor on Koror Island in Palau), but in 1998 were no longer seen there (N. Idechong, Division of Marine Resources, Palau, pers. comm as cited in NMFS and USFWS 1998). Also in the Republic of Palau, dredging and filling for Ollei Dock (Ngerechelong), Ngapeng Dock, and Melekeok Dock were reported to have modified current and sedimentation patterns and degraded or destroyed seagrass, mangrove, and coral reef habitats; and more dock construction was believed to be likely (NMFS and USFWS, 1998).

Marine pollution, including direct contamination and structural habitat degradation, can affect green turtle neritic and oceanic habitat. In the Republic of Palau, environmental contamination in the form of sewage effluent is a problem around Koror State, particularly Malakal Harbor, and near urban areas (NMFS and USFWS, 1998). In the Solomon Islands, sewage discharges from land and discharges of garbage, bilge water, and other pollutants from ships have been identified as sources of pollution to the coastal and marine environments (Solomon Islands Ministry of Environment Conservation and Meteorology, 2008). Land-based activities, including logging, plantation development, and mining, often cause excessive sedimentation of nearshore waters (Sulu et al., 2000). However, the extent to which pollution and sedimentation may be affecting green turtle foraging habitat in the Solomon Islands is unknown.

Environmental contamination was identified as a minor problem in the Marshall Islands in 1998 (NMFS and USFWS, 1998). Some possible problems were identified at Kwajalein, Bikini, and Anewetok, where toxic and hazardous waste were dumped into coastal waters during the era of military missile testing. More recently, Rudrud et al. (2007) found that there is a high probability of green turtles being exposed to toxicants remaining in the Marshall Islands from war and weapons testing (e.g., foraging on algae growing on toxic surfaces, resting near irradiated shipwrecks).

Green turtle foraging areas around Wake Island may be contaminated from an old World War II steel dump as evidenced by the continued presence of algal mats that point to increased iron levels in the water (Defense Environmental Network and Information Exchange, undated). No well-documented records exist on green turtle occurrence in Nauru (Project GloBAL, 2009c), although Buden (2008) refers to several reports that mention the presence of green turtles in Nauru. However, because of low green turtle abundance, any marine pollution that exists (e.g., sewage discharge, small oil spills from barges in the harbor) but is believed to be minor (Jacob, 2000) is likely to have minimal or no impact on green turtles.
Climate change also may result in future trophic changes, including changes in the distribution, amount, and types of seagrasses and macroalgal species (Lapointe, 1999; Harley et al., 2006; Björk et al., 2008), thus altering green turtle foraging habitat (Hawkes et al., 2009).

11.2.5.2. **Factor B: Overutilization**

Overutilization for commercial and subsistence purposes likely was a factor that contributed to the historical declines of this DPS. Current legal and illegal harvest of green turtles for human consumption continues as threat to this DPS.

*Egg Harvest*

One of the most detrimental human threats to green turtles is the intentional harvest of eggs from nesting beaches (NMFS and USFWS 2007). Directed take of eggs is an ongoing problem in the Central West Pacific in the CNMI, Federated States of Micronesia, Guam, Kiribati (Gilbert Islands chain), Papua, Papua New Guinea, Republic of the Marshall Islands, and the Republic of Palau, and the Ogasawara Islands of Japan (Eckert, 1993; Guilbeaux, 2001; Hitipeuw and Maturbongs, 2002; Philip, 2002). In addition to the collection of eggs from nesting beaches, the killing of nesting females continues to threaten the stability of green turtle populations. Ongoing harvest of nesting adults has been documented in the CNMI (Palacios, 2012b), Federated States of Micronesia (Cruse, 2009), Guam (Cummings, 2002), Papua (Hitipeuw and Maturbongs, 2002), Papua New Guinea (Maison et al., 2010), and Republic of Palau (Guilbeaux, 2001). Mortality of turtles in foraging habitats is also problematic for recovery efforts. Ongoing intentional capture of green turtles in their marine habitats has been documented in southern and eastern Papua New Guinea (Limpus et al., 2002) and the Solomon Islands (Broderick, 1998; Pita and Broderick, 2005).

Sea turtles are considered a traditional delicacy for most ethnic groups in the CNMI, and turtles and eggs are readily taken on nesting beaches or in coastal waters (McCoy, 1997; NMFS and USFWS, 1998). Knowledge of existing regulations does not inhibit many people from eating turtles or their eggs. During March-August 2009, 16 green turtle nests (estimated to have been laid by five nesting turtles) were documented during intensive monitoring of seven beaches on Saipan, and three (60 percent) of the five potential nesting turtles, as well as three nests, were illegally harvested (CNMI Division of Fish and Wildlife 2009), suggesting that poaching remains a significant threat to turtles on Saipan (Maison et al., 2010).

*Turtle Harvest*

Turtle harvest is likely the most significant source of mortality within the Federated States of Micronesia (NMFS and USFWS, 1998). In general, both turtles and eggs are consumed if encountered (direct take varies from island to island based on rights to the resource) (NMFS and USFWS, 1998; Buden and Edward, 2001). Nesting has apparently been extirpated from one state (Kosrae) due to overharvest of nesting females. The turtle population at East Fayu was on the verge of extirpation (M. Nelson, Marine Resources Division, Federated States of Micronesia, pers. comm as cited in NMFS and USFWS, 1998), because turtles were sold commercially as
well as taken for subsistence. Very little nesting occurs in Pohnpei, less so than in the past (Buden and Edward, 2001). Important turtle nesting beaches in Chuuk lagoon have all been depleted of turtles, except for one or two islands of the southern barrier reef (NMFS and USFWS, 1998). Turtles and turtle eggs have been exploited in Yap State for as long as people have inhabited these islands, and turtles continue to play a subsistence role and are an important part of traditional culture (Maison et al., 2010). Turtles nesting on or mating in the Ulithi Atoll have traditionally been hunted for their meat and eggs (Lessa, 1984). Due to a lack of good fishing grounds around the island of Falalop (Ruddle, 1996), local people harvest green turtles as a food source. For all other islands within the Ulithi Atoll, nesting populations have been conserved as a result of restrictions placed on cultural harvest by the people of the chief island, Mogmog (Lessa, 1984). These restrictions require that all turtles caught within Ulithi Atoll be taken to Mogmog for ritualistic slaughter and sharing of the turtle meat. While turtle harvest has not been systematically assessed or quantified in Ulithi (Cruce, 2009), turtle harvests may exceed local subsistence levels, with outside trade occurring (Andy Tafileichig pers. comm as cited in Kolinski et al. 2004).

Illegal take of turtles and eggs is also a major threat to green turtles on Guam (NMFS and USFWS, 1998; Project GloBAL, 2009a). Turtles have been traditionally taken by residents for celebrations, and reports indicate that illegal harvesting still occurs (Guam Division of Aquatic and Wildlife Resources, 2011, 2012a, 2012b, 2013).

The consumption of nesting turtles and their eggs is the single most important source of turtle mortality in the Republic of the Marshall Islands (NMFS and USFWS, 1998; McCoy, 2004). The harvest of nesting turtles and their eggs is illegal, but there is little or no control over harvest on any of the islands. The turtles are primarily harvested from the nesting beaches and are generally taken for celebrations. Although harvests consist mostly of nesting turtles and their eggs, turtles are also taken in nearshore waters (McCoy, 2004).

Directed take is also considered to be a major problem in the Republic of Palau (NMFS and USFWS, 1998; Guilbeaux, 2001; Isamu and Guilbeaux, 2002; Eberdong and Klain, 2008). Most nesting beaches occur on inhabited islands (Helen Atoll, Merir, Tobi, Kayangel), and residents of these remote nesting areas have been dependent on green turtles for food. As transportation to these remote areas improves, pressures on turtle populations are bound to increase. Although harvest of turtles in coastal waters is legal during 7 months of the year, take of nesting turtles and eggs is not. However, enforcement is inadequate and violations of regulations are common (Guilbeaux, 2001).

In 2012, five (55.6 percent) of nine females documented as nesting on Saipan beaches were poached (Palacios, 2012b). On Tinian, during a two day rapid nesting beach assessment on July 22 and July 29, 2009, evidence of an illegally harvested nesting female was found (CNMI Division of Fish and Wildlife 2009). In 2012, three (10 percent) of 30 nests documented on Tinian showed evidence of poaching; however, this is likely an underestimate (Wenninger, 2012).

In Kiribati, only the westernmost islands (referred to here as the Gilbert Islands chain) are part of the Central West Pacific DPS. The Gilbert Islands consist of a chain of 16 atolls and 20 coral islands including Tawara, the capital of Kiribati. In Kiribati, the Wildlife Conservation
Ordinance (Laws of the Gilbert Islands, 1977) only fully protects green turtles on certain islands outside the Gilbert Islands chain (Maison et al., 2010). Therefore, green turtle harvest is not regulated in the portion of Kiribati that occurs within this DPS. Historically, green turtles and their eggs have been harvested throughout Kiribati (Groombridge and Luxmoore, 1989). According to Eckert (1993), harvest of foraging and nesting turtles appeared to be widespread and primarily noncommercial, but the full extent of exploitation, trade, and use cannot be determined from published data.

Results of a historical review of marine resources of the Raja Ampat Archipelago, Papua Province, Indonesia, suggest that there has been a 50 percent decline in the sightings of sea turtles, fishes, and invertebrates since the early 1800s, likely due to subsistence and commercial exploitation of marine resources (Palomares et al., 2007). In adjacent Papua New Guinea, the major threat impacting green turtles is overharvest for both meat and eggs (Philip, 2002; Project GloBAL, 2009d).

Based on intensive surveying between 1993 and 1996, Broderick (1998) concluded that the Solomon Islands serve as important developmental habitat for juvenile green turtles, but that a large proportion of turtles were being harvested. Pita and Broderick (2005) estimated that over 1,000 green turtles were being harvested per year in the Solomon Islands in three different villages (Kia, Wagina, Katupika) of Isabelle and Choiseul Provinces. Within the Hele Islands of the Western Province, which are reported to have potentially important green turtle nesting habitat, heavy harvesting pressure on eggs and nesting turtles is believed to be limiting nesting success (Argument et al., 2009). Although closed harvest seasons from June to August and from November to January have been in place since 1993 for all turtle species and turtle eggs, subsistence use of turtles continues (Sulu et al., 2000; Wilson et al., 2004; Solomon Islands Ministry of Environment Conservation and Meteorology, 2008). Thus, overexploitation from subsistence harvest during the open seasons, as well as illegal poaching at other times, continues to threaten green turtles in the Solomon Islands.

Historically, green turtles have been harvested for their meat in the Ogasawara Islands, and records show a rapid decline in the sea turtle population between 1880 and 1920 (Horikoshi et al., 1994; Ishizaki, 2007). By the start of the 20th century, efforts, although unsuccessful, were undertaken to manage sea turtles through harvest regulations and artificial hatcheries (Ishizaki, 2007). Currently, sea turtle harvest is strictly regulated with a harvest limit of 135 mature turtles per year (Ishizaki, 2007).

Another threat affecting green turtles in the Central West Pacific is the harassment of nesting turtles. For instance, in the Ogasawara Islands of Japan, nighttime tourist and resident activity on beaches to view and photograph nesting turtles is a problem, resulting in harassment of nesting turtles and increased aborted nesting attempts (Ishizaki et al., 2011); however, the full extent of these impacts is unknown.

11.2.5.3. **Factor C: Disease or Predation**
Nest and hatchling predation likely was a factor that contributed to the historical decline of this DPS. The best available data suggest that current nest and hatchling predation on several Central West Pacific nesting beaches is a threat to this DPS.

The potential effects of disease and endoparasites also exist for green turtles found in the Central West Pacific Ocean. The loss of eggs to non-human predators is a severe problem in some areas. These predators include domestic animals, such as cats, dogs, and pigs, as well as wild species such as rats, mongoose, birds, monitor lizards, snakes, and crabs, ants, and other invertebrates (NMFS and USFWS, 1998).

In the Federated States of Micronesia, disease is problem with unknown impact. Twelve of 702 (1.7 percent) female green turtles tagged at Gielop Island between 1990 and 1993 had carapace lesions that were diagnosed as fibropapilloma (Kolinski, 1994). Lesions of this type have also been reported on turtles foraging around Yap proper, as well as turtles in the Elato and Lamotrek regions (Kolinski, 1994). More recently, Cruce (2008) reported carapace lesions on four (5.8 percent) of 69 turtles encountered on Loosiep Island, but samples had not yet been analyzed. She reported that the lesions were similar to those observed on Gielop Island during the 2005–2007 nesting seasons, the majority of which were suspected to be burrowing barnacle infestations and one was reported to be a papilloma.

In Yap State in the Federated States of Micronesia, nest predation by ghost crabs was reported to be a substantial problem in the 1990s on Olimaarao Island, as well as a potential threat on Falipi Island, both within the Olimaarao Atoll (S. Kolinski and A. Smith, pers. comm cited in NMFS and USFWS 1998). Also in the 1990s, ghost crabs were identified as nest predators on Gielop Island within the Ulithi Atoll, although the extent of damage was less than that reported in the Olimaarao Atoll (NMFS and USFWS, 1998). No recent information on nest predation by ghost crabs is available. During 2008, monitor lizards were observed to depredate 23 of 28 (83 percent) marked green turtle nests on Loosiep Island (Cruce, 2009). Also during 2008, wild pigs were observed to dig into nests that had been depredated by monitor lizards (Cruce, 2008). Although monitor lizards have been historically reported on Bulbul and Yeew Islands by local property owners, they were not observed during 2008 sea turtle nesting surveys (Cruce, 2009). Monitor lizards have been documented as a predator of green turtle nests on Sorol Island within the Sorol Atoll in Yap State; but the lizard has apparently not spread to other islands within the atoll (Buden, 2013).

Polynesian rat predation on nests is a major threat to green turtles in the Republic of the Marshall Islands. According to a 1992 survey by Puleloa and Kilma (1992 as cited in NMFS and USFWS 1998), Polynesian rat predation is very severe at Bikar. Because of the importance of Bikar (largest nesting area for green turtles in the Marshall Islands) this must be considered extremely serious. Predators have also been documented to consume large numbers of eggs on Erikub Atoll (NMFS and USFWS, 1998). On Guam, nest predation by monitor lizards, wild pigs, rats, and crabs is a problem (Cummings, 2002). Nest predation by wild pigs and monitor lizards is also a threat to green turtles in the Republic of Palau; however, the extent of the problem is unknown. In the Solomon Islands, nest predation by crabs, megapodes, and iguanas is high in the Arnavon Marine Conservation Area, a major green turtle nesting beach in the Solomon...
Islands (Wilson et al., 2004). Predation of turtle nests and hatchlings by dogs and feral pigs has been identified as a problem on Warnandi beach in Papua, Indonesia (Maturbongs, 2000). Within the Ogasawara Islands of Japan, ghost crabs (Ocypode cordimana) were documented to have completed depredated 14 to 23 percent of study nests each season from 1991 through 1994 (Suganuma et al., 1996).

11.2.5.4. Factor D: Inadequacy of Existing Regulatory Mechanisms

Our review of regulatory mechanisms under Factor D demonstrates that although regulatory mechanisms are in place that should address direct and incidental take of Central West Pacific green turtles, these regulatory mechanisms are insufficient or are not being implemented effectively to address the population trajectories of green turtles. We find that the inadequacy of existing regulatory mechanisms for fishery bycatch (Factor E) and impacts to nesting beach habitat (Factor A) continue as threats to this DPS.

A minimum of 15 regional or international instruments apply to the Central West Pacific DPS (see section 11.2.5.1). Hykle (2002) and Tiwari (2002) reviewed the value of some international instruments, which vary in their effectiveness. Often, international treaties 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, 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 a 2002 special issue of the Journal of International Wildlife Law and Policy: International Instruments and Marine Turtle Conservation (Hykle, 2002).

Fishery bycatch that occurs throughout the Central West Pacific (see Factor E), as well as anthropogenic threats to nesting beaches (Factor A) and eggs/hatchlings (Factors A, B, C, and E), is substantial. Although conservation efforts to protect some nesting beaches are underway, more widespread and consistent protection would speed recovery. Although national and international governmental and non-governmental entities in the Central West Pacific are currently working toward reducing green turtle bycatch, it is unlikely that this source of mortality can be meaningfully reduced across the range of the DPS in the near future because of the lack of bycatch reduction in commercial and artisanal fisheries operating within the range of this DPS, the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and lack of availability of comprehensive bycatch reduction technologies.

11.2.5.5. Factor E: Other Natural or Manmade Factors

The Central West Pacific DPS of the green turtle is negatively affected by both natural and manmade impacts as described above in Factor E. Within Factor E, we find that fishery bycatch that occurs throughout the Central West Pacific, particularly bycatch mortality of green turtles from longline, pole and line, and purse seine fisheries, continue as threats to this DPS. In
addition, changes likely to result from climate change and natural disasters are also persistent threats to this DPS.

**Incidental Bycatch in Fishing Gear**

Incidental capture in artisanal and commercial fisheries is a threat to the survival of green turtles in the Central West Pacific. Sea turtles may be caught in longline, pole and line, and purse seine fisheries.

In the Republic of the Marshall Islands, a purse-seine fishery for tuna and a significant longline fishery operate in the EEZ, and sea turtles have been captured in both fisheries with mortality sometimes occurring (Hay and Sablan-Zebedy, 2005). McCoy (2007a) presented a summary of sea turtle interactions with longline vessels based in Majuro from observer data from 2005 to 2007. A total of 33 sea turtle interactions were documented during this period, of which six (18 percent) were identified as green turtles. The mortality rates recorded for these 33 interactions were high, with only five turtles identified as alive upon release (McCoy, 2007a).

In Palau, a total of 18 sea turtles were captured on shallow-set longline vessels during 12 trips with observer coverage from April–December 2007. Out of the 18 interactions, two (11 percent) were green turtles (McCoy, 2007b). One was landed onboard alive and released, the other was dead at the time of landing. The catch per unit effort of the 18 interactions was 0.26 turtles per 1,000 hooks, with an average of 1,442 hooks deployed per 47 sets observed during the 12 trips. Taking into consideration that in February 2007, approximately 100 longline vessels were licensed to fish in the Palau EEZ, with about 50 to 80 actually actively engaged in the fishery in Palau, the potential for interactions with green turtles is relatively high.

In the Federated States of Micronesia EEZ and surrounding areas, an Oceanic Fisheries Programme (2001) review determined that 83 sea turtles were captured in 2,143 observed longline sets from 1990–2000 in an area described as the western tropical Pacific from 10°N to 10°S. McCoy (2003) estimated that the percentage of overall longline effort represented by these 2,143 observed sets was likely less than 2 to 5 percent. The condition of the 83 turtles captured in these sets was identified as 58 percent alive and healthy, 8 percent alive but injured or stressed, 6 percent barely alive, and 27 percent dead (Oceanic Fisheries Programme, 2001). Although green and olive ridley turtles made up the majority of sea turtles that could be identified to the species level, a large number of the turtles encountered could not actually be identified, so the actual species composition of sea turtle interactions in the longline fisheries could not be determined.

In the Solomon Islands, domestic and foreign purse seine and pole and line fisheries, as well as a foreign longline fishery, participated in the commercial tuna fishery in 2007 (Western and Central Pacific Fisheries Commission, 2008). In the CNMI, numerous subsistence and small-scale commercial fishing operations occur along Saipan’s western coast and along both the Rota and Tinian coasts (CNMI Coastal Resources Management Office 2011). Incidental catch of turtles in Guam coastal waters by commercial fishing vessels probably also occurs (NMFS and USFWS, 1998). However, no bycatch studies have been undertaken to quantify the level of incidental capture by commercial fishing operations in the Solomon Islands (Project GloBAL,
2009e), the CNMI (Project GloBAL, 2009b), or Guam (Project GloBAL, 2009a). In 2007, 222 fishing vessels (200 purse-seiners and 22 longliners) had access to Papua New Guinea waters (Kumoru, 2008). Although no official reports have been released on sea turtle bycatch within these fisheries (Project GloBAL, 2009d), sea turtles interactions with both fisheries have been commonly observed (Kumoru, 2008). However, the level of mortality is unknown.

High-seas drift net fishing in the Central West Pacific ended with a United Nations moratorium in December 1992. However, there is virtually no information on the incidental take of sea turtle species by the drift net fisheries in the Central West Pacific prior to the moratorium. The cessation of high-seas drift net fishing in 1992 should have reduced the incidental take of sea turtles. However, nations involved in drift net fishing may have shifted to other gear types; this shift in gear types could have resulted in either similar or increased turtle bycatch and associated mortality.

**Vessel Strikes**

The impacts of vessel strikes in the Central West Pacific is unknown but probably inconsequential, except possibly in the Republic of Palau where high speed skiffs constantly travel throughout the lagoon south of the main islands (NMFS and USFWS, 1998). However, green turtles have been documented as occasionally being hit by boats in Guam. In May 2012, one stranded green turtle with evidence of being hit by a vessel washed ashore east of Kilo Wharf on Naval Base Guam (Guam Division of Aquatic and Wildlife Resources, 2012b). Another green turtle that stranded dead at Uniform Wharf at Naval Base Guam in September 2011 had a gash on the carapace that may have been from a vessel strike (Guam Division of Aquatic and Wildlife Resources, 2012a).

**Pollution**

In the Federated States of Micronesia, debris is dumped freely and frequently off boats and ships (including government ships). Landfill areas are practically nonexistent in the outer islands and have not been addressed adequately on Yap proper or on Chuuk and Pohnpei. The volume of imported goods (including plastic and paper packaging) appears to be increasing. Some people have observed plastic debris in the gut contents of harvested turtles, but the extent of this problem is unknown (NMFS and USFWS, 1998). In the Republic of Palau, entanglement in abandoned fishing nets has been identified as a threat to sea turtles (Eberdong and Klain, 2008).

In the Republic of the Marshall Islands, debris and garbage disposal in coastal waters is a serious problem on Majuro Atoll and Ebete Island (Kwajalein Atoll) both of which have inadequate space, earth cover, and shore protection for sanitary landfills. This problem also exists to a lesser extent at Daliet Atoll (NMFS and USFWS, 1998).

A study of the gastrointestinal tracts of 36 slaughtered green turtles in the Ogasawara Islands of Japan in 2001 revealed the presence of marine debris (e.g., plastic bag pieces, plastic blocks, monofilament lines, Styrofoam pieces) in the majority of the turtles (Sako and Horikoshi, 2003). Eleven of the 36 turtles (30.6 percent) had marine debris in their stomachs, while 25 of the 36
turtles (69.4 percent) had marine debris in their intestines. One turtle had an obstruction in the intestine; most turtles had gastrointestinal tract inflammation.

**Climate Change**

Similar to other areas of the world, climate change and sea level rise have the potential to impact green turtles in the Central West Pacific. Over the long term, Central West Pacific turtle populations could be threatened by the alteration of thermal sand characteristics (from global warming), resulting in the reduction or cessation of male hatching production (Kasperek et al., 2001; Camiñas, 2004; Hawkes et al., 2009; Poloczanska et al., 2009). Further, a significant rise in sea level would restrict green turtle nesting habitat in the Central West Pacific. Coastal erosion has been identified as a high risk in the CNMI due to the existence of concentrated human population centers near erosion-prone zones coupled with the potential increasing threat of erosion from sea level rise (CNMI Coastal Resources Management Office 2011). In the Federated States of Micronesia, Yap State’s low coralline atolls are extremely vulnerable to rises in sea levels and will be adversely affected if hypothesized rises occur (NMFS and USFWS, 1998). These risks are high for all beaches in the Central West Pacific. Interestingly, Barnett and Adger (2003) identified projected increases in sea-surface temperature, and not sea level rise, as the greatest long-term risk of climate change to atoll morphology and thus to atoll countries like those in the Central West Pacific. They state that coral reefs, which are essential to the formation and maintenance of the islets located around the rim of an atoll, are highly sensitive to sudden changes in sea-surface temperature. Thus, climate change impacts could have profound long-term impacts on green turtle nesting in the Central West Pacific, but it is not possible to project the impacts at this point in time.

**Natural Disasters**

Natural environmental events, such as cyclones and hurricanes, may affect green turtles in the Central West Pacific DPS. These storm events have also been shown to cause severe beach erosion and likely have negatively affected hatching success at many green turtle nesting beaches, especially in areas already prone to erosion. Shoreline erosion occurs naturally on many islands in the atolls of the Marshall Islands due to storms, sea level rise from the El Niño–Southern Oscillation, and currents (NMFS and USFWS, 1998). Some erosion of nesting beaches at Oroluk was reported in 1990 after passage of Typhoon Owen (NMFS and USFWS, 1998).

11.2.6. **Summary of Existing Conservation Efforts**

The main threats to Central West Pacific DPS green turtles include fishery bycatch, coastal development and beachfront lighting, sand mining, marine debris and pollution, legal and illegal harvest, and nest and hatching predation. Most Central West Pacific countries have developed national legislation to protect sea turtles and nesting habitats. National protective legislation generally prohibits intentional killing, harassment, possession, trade, or attempts at these; however, a lack of or inadequate enforcement of these laws appears to be pervasive.

At least one country (Palau) has site-specific conservation for sea turtle habitat protection. For example, two nationally mandated protected areas (Ngerukewid Islands Wildlife Preserve and
Ngerumekaol Spawning Area) within Koror State in Palau exist, and restrictions are placed on entry and fishing within established boundaries. While it is important to recognize the success of these protected areas, very few areas that host important green turtle nesting or foraging aggregations have been designated as protected areas within the Central West Pacific. Therefore, existing protected areas may not be sufficient for the conservation of the species within this DPS.

Marine debris is a problem on some green turtle nesting beaches and foraging areas in the Central West Pacific, in particular on the nesting beaches of the CNMI (Palacios, 2012a) and in the nearshore foraging areas of the Federated States of Micronesia, Republic of the Marshall Islands, and Republic of Palau (NMFS and USFWS, 1998; Eberdong and Klain, 2008). Organized beach clean-ups on some CMNI beaches have been conducted to help mitigate this impact (Palacios, 2012a).

Although high-seas drift net fishing in the Central West Pacific ended with a United Nations moratorium in December 1992, and the cessation of high-seas drift net fishing should have reduced the incidental take of sea turtles, it is likely that nations involved in drift net fishing shifted to other gear types that may have resulted in either similar or increased turtle bycatch and associated mortality. Given the lack of conservation efforts to address fisheries and the limited in-water protection provided to turtles to reduce the additional impacts of pollution and marine debris interactions, it is unlikely that the status of the species will change given the measures discussed here.

Overall, it appears that international and national laws to protect green turtles may be insufficient or not implemented effectively to address the needs of green turtles in the Central West Pacific. This minimizes the potential success of existing conservation efforts.

11.2.6.1. **International Instruments**

A minimum of 15 treaties or other regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the Central West Pacific Ocean. The international instruments listed below apply to sea turtles found in the Central West Pacific Ocean. See Appendix 5 for a description of each of these instruments.

- Convention on Biological Diversity
- Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention)
- Convention on the Conservation of Migratory Species of Wild Animals
- Convention on International Trade in Endangered Species of Wild Fauna and Flora
- Convention for the Prohibition of Fishing with Long Drift nets in the South Pacific
- Convention for the Protection of the Natural Resources and Environment of the South Pacific Region
- FAO Technical Consultation on Sea Turtle-Fishery Interactions
- Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding (IOSEA)
- International Convention for the Prevention of Pollution from Ships (MARPOL)
- International Union for the Conservation of Nature
11.2.6.2. National Legislation and Protection

In addition to the international mechanisms, most Central West Pacific countries have developed legislation to protect sea turtles and/or nesting habitats. They are summarized below. However, the overall effectiveness of this country specific legislation is unknown at this time.

Commonwealth of the Northern Mariana Islands

In addition to protection under the U.S. ESA, sea turtles in the CNMI are protected by the Fish, Game and Endangered Species Act (PL 2-51). PL 2-51 establishes a Fish and Wildlife Division and states that the Director of Natural Resources shall determine whether any species shall be designated as threatened or endangered. Green and hawksbill turtles are listed as protected species in the CNMI Hunting Regulations, which prohibit hunting for these species. The CNMI Department of Land and Natural Resources, Division of Fish and Wildlife, is the agency with vested authority and responsibility for the conservation of protected species and enforcement of protected species regulations in CNMI (Maison et al., 2010).

Federated States of Micronesia

Yap State Code prohibits the commercial sale of sea turtle meat and eggs (Yap State Code, http://fsmlaw.org/yap/code/, accessed 3/28/2013). Traditionally, nesting green turtles throughout Ulithi Atoll have been managed and conserved by the imposition of cultural limitations on take for consumption, put in place by leaders of the chief island, Mogmog (Lessa, 1984). In recent years, it appears turtle take has increased due to the degradation of traditional practices although the number of turtles taken annually within Ulithi Atoll has not been assessed or quantified (Cruce, 2009). Chuuk State Code was still in draft form at the time of drafting of this report (Draft Chuuk State Code, http://fsmlaw.org/chuuk/code/, accessed 3/28/2012). According to Pohnpei State laws, there is a minimum size limit for greens (34 inches carapace length) and hawksbills (27 inches carapace length) and closed harvest seasons June 1 to August 31 and December 1 to January 31, with nesting turtles protected and egg collecting prohibited at all times (Buden and Edward 2001; Secretariat of the Pacific Regional Environment Program, 2007, as cited in Maison et al. 2010). Although no nesting has been reported in Kosrae State, state code regulates the take of turtles in water with a minimum size limit for all species of 27 inches carapace length, closed seasons June 1 to August 31 and December 1 to January 31, and prohibition of egg collecting and killing turtles while onshore at all times (Maison et al., 2010). The Federated States of Micronesia is not a participating party to CITES.
**Guam**

In addition to protection under the U.S. ESA, sea turtles are protected by the Endangered Species Act of Guam, which adopts the same definitions and status designations as the U.S. ESA and carries additional penalties for violations at the local government level (Maison et al., 2010). The Guam Department of Agriculture Division of Aquatic and Wildlife Resources (DAWR) is the agency with vested authority and responsibility for the conservation of protected species and enforcement of the ESA of Guam. Other Guam resource agencies, such as the Bureau of Statistics and Plans (BSP), also have specific mandates in relation to sea turtle conservation. The BSP administers the Guam Coastal Management Plan (GCMP) through the Coastal Zone Management Act of 1972 (Guam Public Law 92-583 and Public Law 94-370). The GCMP guides the use, protection, and development of land and ocean resources within Guam’s coastal zone, which includes all non-Federal property and all submerged lands and waters out to 3 nm (5.6 km) from the shoreline (Maison et al., 2010).

**Republic of the Marshall Islands**

The harvest of sea turtles in the Republic of the Marshall Islands is regulated by the Marine Resources Act (1997), which sets minimum size limits for greens (34 inches carapace length) and hawksbills (27 inches carapace length) and closed seasons from June 1 to August 31 and December 1 to January 31 (Maison et al., 2010). Egg collecting and take of turtles while they are onshore is prohibited at all times. The Marshall Islands Marine Resources Authority is the entity with the responsibility of managing marine resources in the Marshall Islands. The Republic of the Marshall Islands is not a participating party to CITES.

**Republic of Palau**

Palau domestic fishing laws (24 PNCA 1201) specify minimum size limits for green turtles (34 inches carapace length) and hawksbills (27 inches carapace length) and closed seasons from June 1 to August 31, and December 1 to January 31 (Secretariat of the Pacific Community and Bureau of Marine Resources Palau, 2007). Taking of eggs or female turtles while onshore is prohibited at all times. In addition, two nationally mandated protected areas (Ngerukewid Islands Wildlife Preserve and Ngerumeakaol Spawning Area) within Koror State provide additional protection to green turtles through restrictions placed on entry and fishing within established boundaries. Nesting habitat, nesting turtles, and eggs are also afforded protection within the Ngerukewid Islands Wildlife Preserve (Guilbeaux, 2001).

Efforts have also been made by some Palau states for the protection of sea turtles, including moratoriums and bans on the take of specific species, take of a particular life stage, and implementation of protected areas (Guilbeaux, 2001). Examples include the creation of the Ngeruangel Reserve Management Plan, which restricts harvest levels and circumstances under which turtles can be harvested from Ngeruangel Atoll in Kayangel State, and the implementation of no-fishing and limited public access areas that offer some protection to turtles in the water, as well as nesting turtles and eggs, in Koror State (Guilbeaux, 2001). However, many of Palau’s
states do not have sufficient funds, infrastructure, and motivation to implement and enforce these initiatives.

**Wake Island**

Wake Island is an unincorporated possession under the U.S. Department of the Interior’s authority, but is currently used and managed by the U.S. Department of Defense. Wake Island does not support resident human populations and does not have a local government; therefore, no local regulations exist to protect sea turtles. However, green turtles at Wake Island are protected by the U.S. ESA. In addition, the U.S. Department of Defense restricts access within a 3-nautical mile Naval Defensive Sea Area at Wake Island, which minimizes the potential for fishing impacts in this area.

**Kiribati**

In Kiribati, the Wildlife Conservation Ordinance (Laws of the Gilbert Islands, 1977) prohibits hunting, killing, or capturing any wild turtle on land and fully protects the green turtle in the following places: Birnie Island, Caroline Island, Christmas Island, Flint Island, Gardner Island (Nikumaroro), Hull Island (Orona), Malden Island, McKean Island, Phoenix Island, Starbuck Island, Sydney Island (Manra), and Vostock Island (Maison et al., 2010). However, none of these islands are within the Gilbert Islands chain, which is the only chain of islands within Kiribati that lie within the Central West Pacific DPS. Kiribati is not a participating party to CITES.

**Nauru**

There is no prohibition or protection for sea turtles in the Nauru Fisheries and Marine Resources Authority Act of 1997 or any other maritime legislation (Project GloBAL, 2009c). Nauru is not a participating party to CITES.

**Papua**

In 1999, the Indonesia Government passed Government Regulation No. 7 for the protection of all turtle species including the green turtle (Zainudin et al., 2008). Papua is not a participating party to CITES.

**Papua New Guinea**

In Papua New Guinea, marine resources and lands are owned by a large number of clan and sub-clan groups whose tenure rights are recognized in the national Constitution (Maison et al., 2010). With respect to sea turtles, the 1976 Fauna (Protection and Control) Act restricts the harvesting of protected wildlife, the devices and methods by which fauna may be taken, and the establishment of localized protective regimes on land and waters under customary tenure (Berdach and Mandeakali, 2004; Kinch, 2006). Additionally, the Paua New Guinea is a member party to CITES and regulates and restricts the export of CITES listed species (http://www.cites.org/eng/disc/parties/alphabet.php). However, in Papua New Guinea, only
leatherback turtles are protected under the Fauna (Protection and Control) Act. The Act does not formally protect green turtles and makes provisions for persons with customary rights to take or kill turtles, but states that turtles cannot be taken, killed, or sold during the months of May through July. Furthermore, the Act stipulates payments for turtles: (a) K20.00 for a turtle less than 60 cm in length; and (b) K30.00 for a turtle of 60 cm or more in length. The Papua New Guinea Department of Environment and Conservation has the authority and responsibility to enforce laws and environmental Acts.

**Solomon Islands**

The Solomon Islands Fisheries Act (1993) regulations prohibit the sale, purchase, or export of sea turtle species or their parts, protect nesting turtles and eggs during the breeding season (June to August and November to January), and contain specific protections for leatherback turtles (Secretariat of the Pacific Regional Environment Program, 2007, as cited in Maison et al. 2010). The Solomon Islands is a participating party to CITES and the Wildlife Protection and Management Act (1998) prohibits the export of five turtle species or their derivative products (greens, hawksbills, loggerheads, olive ridleys, and leatherbacks) (Maison et al., 2010).

**Ogasawara Islands, Japan**

The Ogasawara Islands were designated a National Park by the Japanese government in 1972, and most uninhabited islands have restricted access (Maison et al., 2010). In Japan, there are eight laws and ordinances that regulate (allow via permit) or prohibit actions harmful to sea turtles, such as taking, buying, and selling turtles, their eggs, and any derivative products, or restrict access to nesting beaches. In general, harvest is prohibited but exemptions may be obtained for subsistence use. Based on the Fishing Law and Law of Fisheries Resources Conservation, each prefecture has at least one Area Fishery Adjustment Committee, which regulates local fishing activities. The Ogasawara Area Fishery Adjustment Committee regulates capture of sea turtles and collection of their eggs on the beach. The Ministry of the Environment has jurisdiction over the Natural Park Law. Capture of sea turtles and collection of their eggs are banned under the law in any special protection zones of national parks and quasi-national parks. Many nesting beaches in Ogasawara Islands are designated as a special protection zone. The Law for the Conservation of Endangered Species of Wild Fauna and Flora is the primary law in Japan that intends to conserve endangered species. It prohibits the capture of sea turtles and eggs for sale for all seven species and prohibits domestic assignment or transfer of endangered species listed in CITES. This law was established in accordance with CITES and is enforced by the Japan Ministry of Environment (Maison et al., 2010).

Fishery bycatch that occurs throughout the Central West Pacific (see Factor E), as well as anthropogenic threats to nesting beaches (Factor A) and eggs/hatchlings (Factors A, B, C, and E), is substantial. Although conservation efforts to protect some nesting beaches are underway, more widespread and consistent protection would speed recovery. Although national and international governmental and non-governmental entities in the Central West Pacific are currently working toward reducing green turtle bycatch, it is unlikely that this source of mortality can be meaningfully reduced across the range of the DPS in the near future because of the lack of bycatch reduction in commercial and artisanal fisheries operating within the range of this DPS.
the lack of comprehensive information on fishing distribution and effort, limitations on implementing demonstrated effective conservation measures, geopolitical complexities, limitations on enforcement capacity, and lack of availability of comprehensive bycatch reduction technologies.

11.3. **Assessment of Significant Portion of its Range (SPR)**

The SRT reviewed the information on threats and extinction risk within each DPS to determine whether there were portions of the range of the DPS that might be considered “significant portions of the range” in accordance with the definition of endangered and threatened species and the draft policy that interprets that term (see Section 3.4).

This DPS includes a large geographic area, yet the total abundance within the DPS is small, at approximately 6500 nests. However, the females from the largest nesting site are distributed throughout the DPS, and ongoing threats are fairly uniform. Therefore, the SRT concluded that the need to consider a significant portion of the range does not apply to this DPS.

11.4. **Assessment of Extinction Risk**

For the Status Review Team's assessment of extinction risk for green turtles in the Central West Pacific DPS, there were two separate sets of ranking exercises: One focusing on the importance that each SRT member placed on each of the six critical assessment elements considered for this region (Table 11.3), and a second which reflects the SRT members’ expert opinion about the probability that green turtles would fall into any one of various extinction probability ranges (Table 11.4). See Section 3.3, for details on the six elements and the voting process.

**Table 11.3.** Summary of ranks that reflect the importance placed by each SRT member on the critical assessment elements considered for the Central West Pacific DPS. For Elements 1-4, higher ranks indicate higher risk factors.

<table>
<thead>
<tr>
<th>Critical Assessment Elements</th>
<th>Element 1 (Abundance 1 to 5)</th>
<th>Element 2 (Trends/ Productivity 1 to 5)</th>
<th>Element 3 (Spatial Structure 1 to 5)</th>
<th>Element 4 (Diversity/ Resilience 1 to 5)</th>
<th>Element 5 (Five-Factor Analyses -2 to 0)</th>
<th>Element 6 (Conservation Efforts 0 to 2)</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN RANK</td>
<td>2.50</td>
<td>2.42</td>
<td>2.17</td>
<td>2.17</td>
<td>-1.08</td>
<td>0.67</td>
</tr>
<tr>
<td>SEM</td>
<td>0.26</td>
<td>0.29</td>
<td>0.27</td>
<td>0.24</td>
<td>0.26</td>
<td>0.22</td>
</tr>
<tr>
<td>RANGE</td>
<td>1–4</td>
<td>1–4</td>
<td>1–4</td>
<td>1–3</td>
<td>(-2) –0</td>
<td>0–2</td>
</tr>
</tbody>
</table>

With respect to the important rankings for the six critical assessment elements, nesting abundance was featured relatively high in the risk threshold voting (only roughly 6,551 nesting
females in the DPS); however the combined effects of abundance, trends/productivity, spatial structure, and diversity/resilience affected overall threshold voting. SRT members also generally thought that, on balance, future threats not yet reflected in the nester abundance by the population weighed heavier in their risk assessment voting than did any conservation efforts that may emerge in the future. With respect to the diversity of opinions among the SRT members when considering the six critical assessment elements, there was a wide range in rankings (i.e., voter opinion) for all of the elements.

Table 11.2. Summary of Green Turtle SRT member expert opinion about the probability that the Central West Pacific DPS will reach a critical risk threshold within 100 years. Each SRT member assigned 100 points across the rank categories. The continuum in Row 1 has categories with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1–5%</th>
<th>6–10%</th>
<th>11–20%</th>
<th>21–50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td><strong>MEAN ASSIGNED POINTS</strong></td>
<td>43.25</td>
<td>19.25</td>
<td>13.75</td>
<td>12.50</td>
<td>8.58</td>
<td>2.67</td>
</tr>
<tr>
<td><strong>SEM</strong></td>
<td>10.92</td>
<td>5.19</td>
<td>3.65</td>
<td>4.54</td>
<td>4.08</td>
<td>1.78</td>
</tr>
<tr>
<td><strong>Min</strong></td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td><strong>Max</strong></td>
<td>99</td>
<td>60</td>
<td>40</td>
<td>40</td>
<td>40</td>
<td>20</td>
</tr>
</tbody>
</table>

Of the critical risk threshold categories describing the probability that the Central West Pacific DPS will reach a critical risk threshold within 100 years (Table 11.4), SRT member votes resulted in the greatest point (i.e., probability) designations in the <1% and '1-5%' risk ranges (mean of 43.25 and 19.25 points, respectively). The '>50%' range received the fewest points from SRT members (mean of 2.67), however the sum of the top three categories combined was significant (23.75).

Vote justifications provided for this DPS varied to some degree across SRT members. Some vote justifications mentioned the encouraging positive impact of the Chichijima nesting trend on the DPS; however some members expressed concern with overall relatively small DPS nesting female abundance level. Spatial diversity of nesting was thought to be adequate for some members, less so for others. Relatively minimal concentration (maximum of 22 percent) of nesting at one site considered to help reduce risk, however some concern regarding uncertainty of trends at a number of locations. Most members expressed concerns regarding threats facing the DPS and the negative impacts that they could have on the DPS.

11.5. Synthesis and Integration

This DPS is characterized by a relatively small nesting population spread across a relatively expansive area roughly 2,500 miles wide (Palau to the Marshall Islands) and 2,500 mile long (Ogasawara, Japan to the Solomon Islands). This DPS is dominated by insular nesting. Fifty-one known nesting sites were analyzed; however 16 sites were “unquantified.” Further study of this DPS is necessary to improve our understanding of it.
The limited available information on trends suggests a nesting population decrease in some areas, an increase in the Japan nesting location, and unknown trends in others. The second largest nesting site in this DPS (Chichijima, Japan) shows positive growth. While this site only has approximately 1,301 nesting females, it exhibits an encouraging increasing population growth rate. The dispersed location of nesting sites and lack of concentration of nesting provides a level of habitat use diversity and population resilience which reduces overall extinction risk; however the small size of some of these sites minimizes their contribution to risk reduction. Additionally, extinction risk is increased as a result of threats facing this DPS.

The combined effects of abundance, trends/productivity, spatial structure, and diversity / resilience considered together affected overall extinction risk threshold determinations. Additionally, on balance, future threats not yet experienced by the population weighed heavier in risk assessment than did conservation efforts that may emerge in the future. The SRT’s voting on the likelihood of reaching a critical risk threshold of extinction within 100 years resulted in 43.25 percent of the votes cast in the ‘<1%’ category, 19.25 percent in the ‘1-5%’ category, 13.75 percent in the ‘6-10%’ category, 12.5 percent in the ‘11-20%’ category, 8.58 percent in the ‘21-50%’ category, and 2.67 in the ‘>50%’ category. The sum of the top three risk categories combined was significant, 23.75 percent of the votes suggest that the extinction risk could be as high as 20 percent to over 50 percent.
12. SOUTHWEST PACIFIC DPS [DPS #8]

12.1. DPS Range and Nesting Distribution

Green turtle nesting is widely dispersed throughout the Southwest Pacific Ocean (Figure 12.1). The bulk of this DPS nests within Australia’s Great Barrier Reef World Heritage Area (GBR) and eastern Torres Strait. The northern GBR and Torres Strait support some of the world’s highest concentrations of nesting (Chaloupka et al., 2008). Nesting sites also occur on the Coral Sea Islands, New Caledonia, and Vanuatu. The largest known nesting area for green turtles in New Caledonia is the d’Entrecasteaux atolls, which are located 258 km north of Grande Terre and include Surprise, LeLeixour, Fabre, and Huon Islands (Maison et al., 2010). Vanuatu hosts over 189 nesting sites on 33 islands (Maison et al., 2010).

![Figure 12.1. Nesting distribution of green turtles in the Southwest Pacific DPS. Size of circles indicates nesting estimated nester abundance (see Section 12.2.1). Locations marked with ‘×’ indicate nesting sites lacking abundance information.](image-url)
12.2. Critical Assessment Elements

In the evaluation of extinction risk for green turtles in the Southwest Pacific DPS, the SRT considered six distinct elements, each of which are discussed below: (1) Nesting Abundance, (2) Population Trends (3) Spatial Structure, (4) Diversity / Resilience, (5) Five-Factor Threat Analyses, and (6) Existing Conservation Efforts. See Section 3.3 for additional information on the selection of these six critical assessment elements.

12.2.1. Nesting Abundance

For the Southwest Pacific DPS, we identified 12 total nesting sites, although it should be noted that perhaps more so than in other DPSs, proximate nesting beaches were grouped. It would be possible to split the nesting aggregations into more than 100 different sites, but because many of the most recent estimates (Limpus 2009) are aggregated, we followed this tendency and aggregated nesting within broad regional areas. Nesting occurs at moderate to high levels within the Southwest Pacific DPS (Tables 12.1 and 12.2). Some isolated locations have extremely high nesting activity. The highest nesting densities in this DPS, and perhaps the entire species, are included in the PVA analysis.

The number of turtles nesting in the GBR area of Australia differs widely from year to year and is well correlated with an index of the Southern Oscillation (Limpus and Nicholls, 2000). For example, the estimate of annual nesters at Raine Island during a medium density nesting season is about 25,000 (Limpus, 2009), and in a high density season (1999–2000) the estimate of nesters at Raine Island increases to 78,672 ± 10,586. Heron Island is the index nesting beach for the Southern Great Barrier Reef (sGBR), and nearly every nesting female on Heron Island has been tagged since 1974 (Limpus and Nicholls, 2000). The mean annual nester abundance from varied between 26 and 1,801 during 1999–2004 (Limpus, 2009).

In comparison to Australia, fewer turtles nest in New Caledonia and Vanuatu. In New Caledonia, Pritchard (1994, cited in Maison et al., 2010) described turtles to be abundant or near saturation levels on the following islands, Surprise, LeLeixour, Fabre, and Huon. A 2006 and 2007 survey of over 6,000 km of nesting habitat identified nesting locations hosting an estimated 1,000 – 2,000 green turtles females nesting annually (Maison et al., 2010 citing Limpus et al., in prep). In Vanuatu, hundreds of nesting green turtles have been observed on Malekula Island, Southern Epi Island, Santo and Thion Islands, Tegua and Hiu Islands (Maison et al., 2010).
Table 12.1. Summary of green turtle nesting sites in the Southwest Pacific DPS. Data are organized by country, nesting site, monitoring period, and estimated total nester abundance [((Total Counted Females/Years of Monitoring) x Remigration Interval], and represent only those sites for which there is an estimate of nester abundance. For sites at which data are reported in number of nests, estimates of number of nesters are determined by dividing number of nests by nest frequency. For a list of references for these data, see Appendix 2.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Australia</td>
<td>Raine Island</td>
<td>n/a</td>
<td>25,000</td>
</tr>
<tr>
<td>Australia</td>
<td>No. 8 Sandbank</td>
<td>1997</td>
<td>&gt; 637</td>
</tr>
<tr>
<td>Australia</td>
<td>Bramble Cay</td>
<td>1976, 1977, 1979, 1980</td>
<td>1,660</td>
</tr>
<tr>
<td>Australia</td>
<td>Other nGBR (including Murray Islands, other outer islands, most inner shelf cays, and mainland coast)</td>
<td>1981–1997</td>
<td>&gt; 535</td>
</tr>
<tr>
<td>Australia</td>
<td>Heron Island</td>
<td>1999–2004</td>
<td>4,891</td>
</tr>
<tr>
<td>Australia</td>
<td>Rest of sGBR (primarily Bushy Island, Percy Islands, Bell Cay, Lady Elliott Island, and the mainland coast)</td>
<td>n/a</td>
<td>n/a</td>
</tr>
<tr>
<td>Coral Sea</td>
<td>All sites in Coral Sea</td>
<td>multiple ranges</td>
<td>1,000</td>
</tr>
<tr>
<td>New Caledonia</td>
<td>Huon, Leleizour, Fabre</td>
<td>2007–2011</td>
<td>1,777</td>
</tr>
<tr>
<td>Vanuatu</td>
<td>Bamboo Bay</td>
<td>2006</td>
<td>165</td>
</tr>
</tbody>
</table>

1. Based on Limpus 2009 “In a medium density nesting season, about 25,000 breeding females can be expected to be present off the island in early December.” If non-nesting females do not aggregate in the waters near Raine Island, then the adult female abundance would be approximately 5.35 times higher than this estimate. If turtles nesting in early December represent 60% of the seasonal total (per Limpus et al 2006 citing Hamann et al 1996), and if they only aggregate at Raine Island during the time they are nesting, then it is possible that this estimate only captures 60% of the seasonal total.
2. Calculated from Limpus 2009, as a function of the estimate of Raine Island for a typical year. “The combined early December population estimate for Raine Island + Moulter Cay = 1.6386 x Raine Island December Estimate + 112.” Because this is estimated as a function of Raine Island, any biases in the Raine Island estimate transfer to the Moulter Cay estimate.

3. Calculated from remigration interval (5.78) and annual nesting females (1801, 26, 700, 1060, 240, 1250) from Limpus 2009.

4. Calculated as the sum of track counts during a 2-week index season for 6 years for Wreck Island 1999-2004 and Northwest Island 1998-2003 (Limpus 2009). This updated value of 31,249 nesters is a significant update to the original value of 2,000 that was considered in our structured decision making process to determine extinction risk. While this is a significant increase, the fact that it is a positive change suggests that SRT extinction risk estimates are conservative.


Table 12.2. Green turtle nester abundance distribution among nesting sites in the Southwest Pacific Ocean DPS. Each row of Table 12.1 is represented as a single nesting site in this table.

<table>
<thead>
<tr>
<th>NESTER ABUNDANCE</th>
<th># NESTING SITES DPS 1</th>
</tr>
</thead>
<tbody>
<tr>
<td>n/a</td>
<td>1</td>
</tr>
<tr>
<td>1–10</td>
<td>0</td>
</tr>
<tr>
<td>11–50</td>
<td>0</td>
</tr>
<tr>
<td>51–100</td>
<td>0</td>
</tr>
<tr>
<td>101–500</td>
<td>2</td>
</tr>
<tr>
<td>501–1000</td>
<td>2</td>
</tr>
<tr>
<td>1001–5000</td>
<td>4</td>
</tr>
<tr>
<td>5001–10000</td>
<td>0</td>
</tr>
<tr>
<td>10001–100000</td>
<td>3</td>
</tr>
<tr>
<td>&gt;100,000</td>
<td>0</td>
</tr>
<tr>
<td><strong>TOTAL SITES</strong></td>
<td><strong>12</strong></td>
</tr>
<tr>
<td><strong>TOTAL ABUNDANCE</strong></td>
<td><strong>83,058</strong></td>
</tr>
</tbody>
</table>

PERCENTAGE at LARGEST NESTING SITE (nGBR, Australia) 38%

Comment [A22]: this table needs a lot of attention. Much of it does not seem to match the data. This could be because Limpus 2009 is a review. Go back to some of the primary literature. E.g. Limpus et al 1984

Response: In general, we did give a lot of deference to Limpus 2009 because it was recent, comprehensive, and is largely consistent with the population information listed in conjunction with Australia’s EPBC.

Comment [A23]: which is this one. I would have thought there are many with no or few abundance data – most would probably fit the 1-10.

Response: This is the remainder of nGBR. Indeed green turtle nesting is widespread throughout this region. However, without specific data and clarity on specific sites, we elected to focus largely on the major nesting sites, that would likely have most influence on consideration about extinction risk for green turtles in this DPS.

Comment [A24]: this is not correct. There are dozens that would fit this.

Response: Green turtle nesting is widespread throughout this region. However, without specific data and clarity on specific sites, we elected to focus largely on the major nesting sites, that would likely have most influence on consideration about extinction risk for green turtles in this DPS. We have requested additional data from the reviewer but the data were not received in time to include in the current report.

Comment [A25]: What are these?

Response: The four sites with 1001-5000 nesters are listed in table 12.1.

Comment [A26]: Which sites have you used? There are many more than 12 with data.

Response: The sites summarized in this table are presented in table 12.1.

Comment [A27]: In Text Reviewer comments: In section 12.5 you quote Raine having an abundance of 50000. These numbers don’t match. Which sites are included (and which sites are merged)? There are well over 100 sites in the GBR and coral sea and while many have few data there are at least 14 with several years of data (7 in nGBR, 5 in SGRR and 3 in the Coral Sea). There are also dozens of sites that would fit the 1-10 category – how have they been dealt with? How come that is scored 0? I can’t see any evidence in the literature of 4 sites with 1000 to 5000 (only BC and NW) so are these merged sites? If so be clear. I am not confident that you have used the...
12.2.2. Population Trends

Nesting occurs in many islands throughout the Southwest Pacific DPS, but there are only two nesting areas (Raine Island and Heron Island, described in more detail below) with long-term (>15 years) annual indices of nesting abundance. For a list of references on trend data, see Appendix 3.

The Raine Island, Australia index count ([1994–2004] intermittent) has high inter-annual variability and a slightly increasing linear trend. Heron Island, Australia, index count (1967–2004, intermittent) also has high interannual variability and a slightly increasing linear trend. Although long robust time series are not available for New Caledonia, recent and historic accounts do not suggest a significant decline in abundance of green turtles nesting in New Caledonia (Maison et al., 2010). The trend at Vanuatu has not been documented (Maison et al., 2010).

The Raine Island (nGBR) nesting index is the mean number of females ashore for nesting (during the first 2 weeks of December) that are counted during one survey of the nesting habitat per night (Limpus, 2009). The number of nesters observed on nightly tally counts was relatively low from 1975 through the early 1980s, then had higher peaks starting in 1984 (Limpus 2009). From the mid-1990s to the mid-2000s, there has been a leveling off of the rate of increase (Chaloupka et al., 2008a).

The Heron Island, Australia, index count is derived from a tagging census of the total annual nesting population. There was a 3 percent per year increase in annual nesting abundance in the subset of data from 1974–1998 (Chaloupka and Limpus, 2001) and a similar 3.8 percent per year increase from the 1974–2002 subset (Chaloupka et al., 2008a). When including all years from 1967–2004 there is an increasing linear trend in the annual nesting population size, but the relationship was not significant (Limpus, 2009). The increase in annual nesting females at Heron Island is concurrent with an estimated increase of 11 percent per year from 1985–1992 for the green turtle foraging population (immature and mature females and males) in Heron Reef/Wistari Reef complex (Chaloupka and Limpus, 2001).

PVAs were one aspect of the Population Trend element and were conducted for nesting sites that had a minimum of 15 years of recent nesting data with an annual nesting level of more than 10 females (for more on data quantity and quality standards used, see Section 3.2). To assist in interpreting these PVAs, we indicate the probability of green turtle nesting populations declining to two different biological reference points, one using a trend-based and the other an abundance-based threshold. The trend-based reference point for evaluating population forecasts is half of the last observed abundance value, i.e., a 50% decline taken from the most recent annual survey. The abundance-based reference point was a total adult female abundance of 300 females (i.e., 100 females per year at a nesting site with a 3-yr female nesting remigration interval). Risk is calculated as the percentage of model runs that fall below these reference points within 100 years. This PVA modeling has important limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or future threats that have not yet impacted the population. It assumes all environmental and anthropogenic pressures...
will remain constant in the forecast period and it relies on nesting data alone. For a full discussion of these PVAs and these reference points, see Section 3.2.

A subset of the trend data was used in the PVA analysis (Figure 12.2 and 12.3). The Raine Island analysis was completed using the average number of turtles observed ashore during one walk around the island in 25 seasons (1976–1982, 1984–1989, 1991–2001, 2004) based on data from Limpus et al. (2007) and Chaloupka et al. (2008a). Caution must be used when interpreting these results because they only represent females observed during one sampling bout on one night versus an accumulation of all females from the whole season. Nesting beach monitoring data indicate that there is a 9.1 percent probability that this population will fall below the trend reference point (50% decline) at the end of 100 years, and a 0.4 percent probability that the Raine Island nesting population falls below the absolute abundance reference (100 females per year) at the end of 100 years.

Figure 12.2. Stochastic Exponential Growth (SEG) Model Output for Raine Island, Australia. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, gray-green dotted line is trend reference, and red dotted line is absolute abundance reference. The units for the Raine Island indices are expressed as adult females, so no transformation from nests to nesters was needed.

The Heron Island analysis was completed using an index of adult female nesters across 31 seasons from 1974–2004 based on data from (Chaloupka et al., 2008a; Limpus, 2009). The units for the Heron Island indices are expressed as adult females, so no transformation from nests to nesters was needed.
Nesting beach monitoring data indicate that there is a 17.5 percent probability that the magnitude of adult females associated with Heron Island nesting will fall below the trend reference point (50 percent decline) at the end of 100 years, and an 8.3 percent probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years. It should be noted, however, that this PVA modeling has important limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. For a full discussion of these PVAs and these reference points, see Section 3.2.

![Figure 12.3. Stochastic Exponential Growth (SEG) Model Output (Figure 6) for Heron Island, Australia. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, gray-green dotted line is trend reference, and red dotted line is absolute abundance reference.](image)

The Raine and Heron Islands nesting indices do not fully describe the productivity of this DPS as there is important ecological and demographic information that is not captured in the nesting index. There was a significant decrease in the late 1990s and early 2000s in the mean carapace size (CCL) of nesting females at Raine Island and Heron Island (Limpus et al., 2002, 2003, 2007; Limpus, 2009). Although this decrease is only a few centimeters or less, it could indicate important population-level changes including disproportionate adult mortality (including possible over harvest which could result in a declining population), several strong year-classes beginning to nest (possibly resulting in an increasing population), changes in mean size of nesting group, or changes in maturation time.

Nesters at Raine Island show an increase in the mean observed remigration interval (Limpus et al., 2002), though it is important to note that observed remigration intervals are influenced by

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Comment [A36]: I heard from col that this has changed – you should get confirmation.
Response: we have written to Dr. Limpus but had not received the updated information by the time this report was compiled.
Tagging effort in previous years. Given that the remigration interval of females returning for only their second season is longer than that for turtles that have nested during multiple prior seasons (i.e., older turtles), the observed increase in mean remigration interval further supports the notion that fewer large turtles are present in the population (Limpus et al., 2002). The decrease in size of nesters at Raine Island coupled with the pattern of increase in remigration intervals is consistent with a reduction of older turtles in the population and maybe an early warning that the Raine Island nesting population may be in the early stages of decline (Limpus et al., 2002).

There are additional concerns about the long-term health of the Raine Island nesting population (Limpus et al., 2003). Total productivity is limited by reduced nesting and hatching success, which at Raine Island appears to be depressed due to habitat issues. For Raine Island, mean nesting success (i.e., probability that a clutch will be laid when a turtle comes ashore for a nesting attempt) can be as low as 3.3 percent (range=1.72 to 4.88, n=2; see Table 7 of Limpus et al., 2006). Reduced recruitment can be caused by flooding of egg chambers by ground water, dry collapsing sand around egg chambers, and underlying rock which prevents appropriately deep egg chambers (Limpus et al., 2003). Death of nesting females occurs at Raine Island. Nightly mortality ranges from 0 to over 70 per night and is highest when nesting the previous night exceeds 1,000 (Limpus et al. 2003).

12.2.3. Spatial Structure

When examining spatial structure for the Southwest Pacific DPS, the SRT examined three lines of evidence including genetic data, flipper and satellite tagging, and demographic data.

Genetic sampling in the Southwest Pacific DPS has been extensive for larger nesting sites along the Great Barrier Reef, the Coral Sea, and New Caledonia; however, there are several smaller nesting sites in this region that still need to be sampled (e.g., Solomon Islands, Vanuatu, Tuvalu, Papua New Guinea). Within this DPS there is significant population substructuring (pairwise FST 0.09–0.79, p<0.05). Of the ten nesting sites studied, four regional genetic stocks have been identified in the Southwest Pacific Ocean; northern Great Barrier Reef, southern Great Barrier Reef, Coral Sea (Dethmers et al., 2006; Jensen, 2010), and New Caledonia (Dethmers et al., 2006; P. Dutton, NMFS, unpublished data). Mixed stock analysis of foraging grounds show that green turtles from multiple nesting beach origins commonly mix in foraging grounds along the Great Barrier Reef and Torres Strait regions (Jensen, 2010), but with the vast majority originating from nesting sites within the Great Barrier Reef. There is evidence of low frequency contribution from nesting sites outside the DPS at some foraging areas.

Nesting beach monitoring along with flipper and satellite tagging show the spatial structure of this DPS is largely consistent with viable populations. Foraging is widely dispersed throughout this DPS and also into other DPS's (Limpus, 2009). Nesting is widely dispersed throughout the region; there is more than one major nesting site; there is evidence of some connectivity between nesting sites within each of the four regional stocks but no connectivity among regional stock, and there is nesting on the continental and on islands. The habitat which hosts most of the documented nesting in this DPS is protected (Limpus, 2009).
Demographic information for nesting turtles is widely available for nesting beaches in the Australian component of the DPS. The following demographic data are provided by Limpus (2009). For the northern GBR stock nesters at Raine Island average 106 cm CCL (n=20,947) in length, have a 12 day re-nesting interval (n=16), 5.3 year remigration interval (n=2,094), and at nearby Bramble Cay [same stock] nesters on average lay 6.2 clutches per season (n=684). Furthermore, green turtle clutches at Raine Island average 104 eggs (n=501) and have an emergence success of 78.2 percent. For the southern GBR stock nesters at Heron Island average 107 cm CCL (n=1,942) in length, have a 14 day re-nesting interval (n=264), 5.8 year remigration interval (n=518), and on average lay 5.1 clutches per season (n=878). Green turtle clutches at Heron Island average 114 eggs (n=85) and have an emergence success of 89 percent.

Growth rates obtained from nearshore capture-mark-recapture studies suggest the Southern Great Barrier Reef subpopulation attains maturity at 30–40 years (Limpus and Chaloupka, 1997; Chaloupka et al., 2004). Stage-based survivorship rates are also available from nearshore studies in sGBR foraging areas. Annual survival was 88 percent for juveniles, 85 percent for subadults, and 95 percent for adults (Chaloupka and Limpus, 2005). The high estimate of adult survival should be viewed with caution given a long-term decline in average nester size and increase in remigration interval (Limpus, 2009) which could be caused by disproportionally high mortality in adult stage classes.

12.2.4. Diversity / Resilience

The components considered under this critical element include the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation (e.g. high vs. low beach face, insular vs. continental nesting sites), and the genetic diversity within the DPS. These are important considerations for assessing the potential impact of catastrophic events such as storms, sea level rise, and disease.

This region has high genetic diversity. It is characterized by high nucleotide diversity resulting from a mix of highly divergent lineages found at nesting sites, some of which are among the oldest lineages found in C. mydas.

Nesting and foraging in this region are relatively diverse for green turtles. Nesting is widely dispersed throughout the region and nesting is not completely limited to islands. Nesting, however, is not evenly distributed throughout the DPS, and some of the densest nesting occurs on Raine Island, which has important habitat-based threats. The pivotal temperature for hatchling sex ratio varies within this DPS, with some nesting sites producing primarily females and some producing primarily males [Limpus, 2009; Fuentes et al. 2009]. Nesting can occur year-round in the most northerly rookeries, but a distinct peak occurs in late December to early January for all Australian rookeries. Foraging occurs year-round and in diverse areas geographically and ecologically (coral and rocky reefs, seagrass meadows and algal turfs on sand and mud flats).

In a study of the nGBR nesting assemblages, none were found to pass a threshold for being vulnerable to cyclonic activity (which overlaps with the main nesting season); two were

Comment [A41]: Yes they do? Michael Jensen, one of your SRT, did the models for the nGBR green turtles.
Response: These models are not published and were only under development and not available at the time of the SRT. They should not be included here.

Comment [A42]: But in the nGBR most is on Raine Island which has significant issues.
Response: Text revised.

Comment [A43]: Very few continental islands. In fact in the nGBR there is only 1
Response: Text revised.

Comment [A44]: Fuentes et al (200) JEMBE
Response: Text revised.
vulnerable to sea level rise, and almost all sites in the study were expected to be vulnerable to increased temperatures by 2070 (Fuentes et al., 2011).

12.2.5. Analysis of Factors Listed Under ESA Section 4(a)(1)

Section 4(a)(1) of the ESA lists five factors (threats) that must be considered when determining the status of a species. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

The following information on these factors /threats pertains to green turtles found in the Southwest Pacific DPS.

12.2.5.1. Factor A: Destruction or Modification of Habitat or Range

Groundwater intrusion and sea level rise affect hatchlings and nesting turtles on high density beaches within this DPS. All in-water life stages of green turtles in this DPS are also affected by fishery practices, channel dredging, and marine pollution, although the extent and level of that effect is not known within this DPS. Coastal development and beachfront lighting also impact green turtles in this DPS.

Terrestrial Zone

Destruction and modification of green turtle nesting habitat in the Southwest Pacific DPS results from beach erosion, beach pollution, removal of native vegetation, and planting of non-native vegetation and natural change (Limpus, 2009). Coastal development and construction, placement of erosion control structures and other barriers to nesting, beachfront lighting, and vehicular traffic minimally impact green turtles in this DPS (Limpus, 2009). Most of the nests at the documented nesting sites within this DPS occur within the protected habitat, but there is still concern about the viability of nesting habitat (Limpus, 2009). Hatching production at Raine Island appears to be reduced due to habitat conditions. In the 1996 to 1997 breeding season, for example, flooding of nests caused a near total loss of viable eggs, and flooding has been a regular event in subsequent years (Limpus et al., 2003; Limpus, 2009). Understanding the root cause of changes to Raine Island nesting habitat is challenging and is the aim of several Australian and State Government research and monitoring projects. These habitat-based threats (particularly related to hatching production) constitute serious threats to this DPS, given the large abundance in the nGBR.

Light disorientation affects hatchlings within the Southwest Pacific DPS**. Green turtle hatchlings are attracted to low pressure sodium vapor lights that are not attractive tologgerhead hatchlings (Limpus, 2009). Between 1993 and 2010, artificial light levels have increased significantly for green turtles in minor rookeries of the nGBR and remained relatively constant.
for the mainland of Australia (part of sGBR) south of Gladstone (Komrowski et al 2014). Exposure to artificial light is low or non-existent at Raine Island, Moulter Cay, Sandbanks 7 and 8, Capricorn-Bunker Islands, Swain Reefs, Bushy Island, and Percy Island Group (Komrowski et al 2014), though difficulty in ocean-finding may be related to topography as well as light sources (Limpus and Kamrowski 2013). Disoriented hatchlings have been found at Heron Island and the camping areas of the Capricorn-Bunker islands (EPA Queensland Turtle Conservation Project, unpubl. data). At Raine Island, lighting can trap hatchlings inshore, resulting in increased predation in inshore waters (Limpus et al., 2003), but a recent study (Komrowski et al 2014) did not detect artificial light on or near the island. Light disorientation may also affect production if nesting females seek dark beaches even if the conditions may be sub-optimal for egg incubation and hatching success (Salmon et al., 2000).

** Based on the reviewer comments and associated updates to the text, artificial lighting is somewhat of a larger problem than initially considered by the SRT during its structured decision making efforts to determine extinction risk.

**Neritic/Oceanic Zones**

Threats to habitat in the neritic and/or oceanic zones include fishing practices, channel dredging, and marine pollution. These threats also occur in the Southwest Pacific DPS, though the internesting habitat adjacent to the nesting sites with the highest documented nesting levels in this DPS is protected by the Great Barrier Reef Coastal Marine Park and the adjacent Great Barrier Reef Marine Park (Limpus, 2009). Protection for marine turtles in the Great Barrier Reef World Heritage area has been increasing since the mid 1990s (Dryden et al. 2008).

12.2.5.2. **Factor B: Overutilization**

Consumption of eggs and nesting turtles by indigenous peoples occur on a portion of the high density nesting beaches within this DPS. Turtle harvesting generally occurs extensively throughout the DPS and is a threat to this DPS.

**Egg Harvest**

The Australian Native Title Act (1993) gives indigenous people a legal right to hunt sea turtles in Australia for traditional, communal, non-commercial purposes (Limpus, 2009). Although the current magnitude of Indigenous harvest is not well-documented, both turtles and eggs have been harvested in the recent past. Egg harvest is likely low in the sGBR because the nesting islands are uninhabited, and most egg harvest in the nGBR likely occurs in the Torres Strait, as opposed to Raine Island and neighboring nesting sites in the GBR (Limpus, 2009).

**Turtle Harvest**

Because turtles that nest in the Southwest Pacific DPS may forage within other DPS boundaries, Southwest Pacific DPS turtles are vulnerable to harvest throughout Australia and neighboring countries such as New Caledonia, Fiji, Vanuatu, Papua New Guinea, Indonesia (Limpus, 2009). Cumulative annual harvest of green turtles that nest in Australia may be in the tens of thousands,
and it appears likely that historic Native harvest may have been in the same order of magnitude (Limpus, 2009). Annual harvest in the southern East Coast is estimated to be 500–1000, mostly large females (Limpus, 2009). Estimated annual harvest is even larger for the northeast part of the country, primarily Torres Strait which may have a large harvest turtles (4,000) and eggs (unquantified). Annual Indigenous harvest from northern and Western Australia may be several thousand turtles per year plus non-permitted egg harvest (Limpus, 2009). Harvest from neighboring countries (New Caledonia, Fiji, Vanuatu, Papua New Guinea, Indonesia) may be as high as several thousand juvenile or adults per year and may threaten the sustainability of the Australian green turtle stocks (Limpus, 2009). Modeling suggests that the population trajectory for the sGBR stock is sensitive to removals of large turtles, consistent with subsistence harvesting (Chaloupka, 2002). The nGBR stock has less precise data and lacks comprehensive modeling of the population, but it is presumed that the nGBR stock is more threatened by harvest than the sGBR stock (Limpus, 2009).

Although there is currently no legal commercial harvest in Australia, there has been intense harvesting in some areas within the last century. The north Australian nesting sites historically had a low intensity of sporadic harvest, but in the summer of 1959 there was a harvest of approximately 1,200 nesting females from Raine Island and Moulter Cay (Limpus et al., 2003).

Aside from this intense, short-lived commercial harvest, the nGBR nesting sites (which support the largest documented abundance within this DPS) appear to be relatively undisturbed by large-scale sustained commercial harvest (Limpus et al., 2003). In contrast, intense green turtle commercial harvest (sometimes exceeding a thousand turtles per year) in the south GBR nesting sites was longer and occurred intermittently for the first 50 years of the twentieth century (Limpus, 2009). Also, in Torres Strait there is a treaty (the Torres Strait Treaty 1985) that allows PNG people to catch turtles within a shared fishing zone. In PNG the take of turtles can be commercial. This is a contentious issue.

12.2.5.3. Factor C: Disease or Predation

Disease and current nest and hatchling predation on several Southwest Pacific nesting beaches is likely a factor that negatively affects this DPS while the best available data suggest that it is a continued threat to this DPS.

The body condition of green sea turtles also appears to be directly affected by sporadic sea grass diebacks (including an important dieback in the 1970s) in the Torres Strait (and Kwan 2008).

The potential effects of diseases and endoparasites, as described in for other DPSs, also exist for green turtles found in the Southwest Pacific DPS. Low levels of fibropapilloma-associated turtle herpesvirus is common in green turtles in some but not all semi-enclosed waters like Moreton Bay and Repulse Bay in Australia, more infrequent in nearshore open waters and rare in offshore coral reef habitats (Limpus, 2009). Mortality and recovery rates from this virus are not quantified but stranded, infected turtles are regularly encountered in south Queensland (Limpus, 2009).
Other health conditions such as coccidiosis, parasites, fungal infections also occur (Limpus, 2009). In late 1991, at least 70 green turtles died from coccidiosis infection in southeastern Queensland, but coccidiosis does not appear to be a static threat given that comparable studies in 1992 and 1993 did not detect the disease (Limpus, 2009). Mortality from parasitic worms is not well quantified, but stranded turtles that are heavily infected with blood flukes are regularly encountered in south and north Queensland; and blood flukes and spirochiid trematodes likely cause some green turtle mortality (Limpus, 2009). Fungi have been identified in association with green turtles (in cloaca and near nests) and are thought to cause the death of some eggs within the nest (Limpus, 2009).

Primary hatchling and egg predators include crabs, birds, fish, and mammals. The magnitude of egg predation is not well documented, but within Australia the highest levels of vertebrate predation on eggs appear to occur within other DPSs or for other species (primarily loggerheads) (Environment Australia, 2003). In Vanuatu, nest predation by feral dogs is a primary threat (Maison et al., 2010). Survivorship of hatchlings in sGBR during the transition from nest to sea (accounting for crab and bird predation) may be quite high (0.98; Limpus, 1971), but survivorship of hatchlings as they (0.4; Gyuris, 1994 as cited in Limpus, 2009) transition across the reef flat from the water’s edge to deep water is likely considerably lower. Similar survivorship estimates are not available for the nGBR, but survival during the nest to sea transition are expected to be low and variable, depending on the predator assemblage. Although many birds co-occur with sea turtle hatchlings in the nGBR, only some birds like the rufous night heron are important predators (Limpus et al., 2003). Terrestrial crabs which occur throughout the nGBR have been observed feeding on turtle hatchlings and eggs, but the crabs are generally of low density (Limpus et al., 2003). Shark predation on hatchlings as well as adults has been documented (Limpus et al., 2003).

12.2.5.4. Factor D: Inadequacy of Existing Regulatory Mechanisms

Regulatory mechanisms are in place that should address direct and incidental take of green turtles in the Southwest Pacific DPS; however, these regulatory mechanisms may not be sufficient or may not be being implemented effectively so as to maximize the recovery potential of green turtles in this DPS. The inadequacy of existing regulatory mechanisms for impacts tonesting beach habitat (Factor A) and overutilization (Factor C) are continued threats to this DPS. In the following section (Factor E), we describe the insufficiency of regulatory mechanisms in relation to several threats including incidental bycatch in fishing gear, boat strikes, port dredging, debris, national defense, toxic compounds, and climate change.

There are a minimum of 16 national and international treaties and/or regulatory mechanisms that pertain to the Southwest Pacific Ocean (see Conservation Efforts below), and the vast majority of green turtles nesting in the Southwest Pacific Ocean have at least some level of protection. Hykle (2002) and Tiwari (2002) reviewed the value of some international instruments, which vary in their effectiveness. Often, international treaties 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, 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

Intentional harvest of green turtles occurs throughout the Southwest Pacific DPS. In Australia, Aboriginal and Torres Strait Islanders, as recognized under the Australian Government’s Native Title Act of 1993, have hunting rights. Indigenous groups, governments, wildlife managers and scientists work together with the aim of sustainably managing turtle resources (Maison et al., 2010 citing K. Dobbs, Queensland Parks Authority, pers. comm., 2010), though traditional harvest remains a threat to green turtle populations. Despite the existing regulatory mechanisms, threats to nesting beaches, eggs, hatchlings, juveniles, and adults through harvest and incidental harm occur throughout the Southwest Pacific DPS.

12.2.5.5. Factor E: Other Natural or Manmade Factors Affecting Its Continued Existence

The Southwest Pacific DPS of the green turtle is negatively affected by both natural and manmade impacts. Fishery bycatch that occurs throughout the Southwest Pacific, particularly bycatch mortality of green turtles from pelagic longline, drift nets, set net, and trawl fisheries, is a continued threat to this DPS. Additional threats from boat strikes, marine pollution, and changes likely to result from climate change, and cyclonic storm events will negatively impact this DPS.

**Incidental Bycatch in Fishing Gear**

Incidental capture in artisanal and commercial fisheries is a threat to the survival of green turtles in the Southwest Pacific Ocean. The primary gear types involved in these interactions include trawl fisheries, longlines, drift nets, and set nets. These are employed by both artisanal and industrial fleets, and target a wide variety of species including prawns, crabs, sardines, and large pelagic fish.

Turtles nesting in the Southwest Pacific DPS are vulnerable (but see section below on TED use) to the Queensland East Coast Trawl Fisheries and the Torres Strait Prawn Fishery, and to the extent they forage west of Torres Strait, they are also vulnerable to the NPF. Total mortality of Australian green turtles in fisheries bycatch is not known because there is not reliable reporting of threatened species bycatch in Australian commercial fisheries (Limpus, 2009). Australian trawl fisheries have increased the number of boats, the length of the shot-times and the number and size of nets towed since the 1960s, but the capture of green turtles has been less frequently reported in prawn trawls in Queensland than loggerheads (Limpus, 2009).

The following summarizes information (Limpus, 2009) about the interaction of green turtles and trawling in Australian waters: In Australia’s prawn fishery, green turtles comprise a relatively small but variable percent (28 percent in the Queensland East Coast Trawl Fisheries, 21 percent in the Torres Strait Prawn Fishery, and less than 10 percent in the NPF) of the total turtle catch.
Observed green turtle bycatch includes all sizes from small immature to adults. The rough order of magnitude of green turtle bycatch may be about one in 10,000 trawls (based on catch rates of 0.0042 and 0.0036 turtles of all species per trawl in the NPF). Reported mortality rates for green turtles are less than 20 percent (which is lower than estimated for loggerhead and hawksbill sea turtles). The total mortality from eastern and NPF is estimated to be 50–100 green turtles per year from the late 1980s to the late 1990s. Turtle excluder devices (TEDs) have been required in most of Australia’s prawn fisheries since 2002 or earlier (NPF since April 2000, East Coast Trawl Fisheries since December 2000, Torres Strait Prawn Fishery since March 2002, and Western Australian prawn and scallop trawl fisheries since 2002). Turtle excluder devices are thought to reduce turtle captures in NPF by two orders of magnitude. The reported number of all species of turtles caught in the NPF is 883 in 1999, 68 in 2000, 113 in 2001, 27 in 2002 and 2003 (Australian Government; http://www.environment.gov.au/soe/2006/publications/drs/indicator/133/index.html#issuesforwhichthisisanindicatorandwhy).

The use of TEDs in the NPF became mandatory, due in part to several factors: (1) Objectives of the Australian Recovery Plan for Marine Turtles, (2) requirement of the Australian Environment Protection and Biodiversity Conservation Act for Commonwealth fisheries to become ecologically sustainable, and (3) the 1996 U.S. import embargo on wild-caught prawns taken in a fishery without adequate turtle bycatch management practices (Robins et al., 2002).

Australian and international longline fisheries capture marine turtles. Precise estimates of international capture of Southwest Pacific Ocean DPS green turtles by the international longline fleet are not available, but they are thought to be larger than the Australian component (DEWHA, 2010). Turtle bycatch by the Eastern Tuna and Billfish Fishery of Australia has been dominated by green and leatherback turtles, the vast majority of which are released alive (81 percent in 2006, and 88 percent in 2007; DEWHA, 2010). Average annual bycatch of all species of turtles within Australia’s Eastern Tuna and Billfish fishery is 42 from 1997 through 2004 (http://www.environment.gov.au/soe/2006/publications/drs/indicator/133/index.html#issuesforwhichthisisanindicatorandwhy) and 16.5 in 2006 and 2008; turtle interactions in the Western Tuna and Billfish Fishery is lower (DEWHA, 2010). To assess the impact of bycatch by integrating information on bycatch rates, mortality rates, and body sizes, Wallace et al. (2013) assigned a bycatch impact score to Regional Management Units for various fisheries. For longline fisheries in the Southwest Pacific Regional Management Unit, they assigned a relatively low bycatch impact score of 1.17 (Wallace et al., 2013).

In addition to threats from prawn trawls fisheries, green turtles may be threatened by other fishing gear (summarized from Limpus, 2009). Although tunnel nets capture many green turtles, they do not appear to have substantial mortality rates. Gill nets (targeting barramundi, salmon, mackerel, and shark) in Queensland and the Northern Territory have been observed to catch green turtles, but the magnitude has not been quantified. Crab pots and float lines entangle green turtles, and although the magnitude of mortality is not quantified, it is presumed to be in the tens per year. Untended “ghost” fishing gear that has been intentionally discarded or lost due to weather conditions may entangle and kill many hundreds of green turtles annually.
Shark Control Programs

Green turtles are captured in shark control programs, but protocols are in place to reduce the impact. The Queensland Shark Control Program is managed by the Queensland Department of Primary Industries and Fisheries (Limpus, 2009) and has been operating since 1962 (Gribble et al., 1998). In 1992, their operations began to be modified to reduce mortality of non-target species (Gribble et al., 1998). The average yearly capture of all species of turtles from 1962–1995 was 119.4 turtles per year, with ≥ 35 percent released alive (59 percent were released with undocumented condition, Gribble et al., 1998). After conservation measures were implemented, the average yearly capture of all turtles from 1992–96 was 84 turtles per year, with 87 percent released alive (Gribble et al., 1998). Hence, immediately following the implementation of conservation measures, the mortality for all turtles within this program (including green turtles) was about 11 turtles per year. Observed green turtle annual mortality during 1998–2003 was 2.7 per year (Limpus, 2009). Green turtles have been captured in the New South Wales shark-meshing program since 1937, but total capture for all turtle species from 1950 through 1993 is roughly 5 or fewer turtles per year (Krogh and Reid, 1996). Post release survival does not appear to have been monitored in any of the monitoring programs.

Boat strikes, Port Dredging, and Military Activities

Other threats such as boat strikes, port dredging, debris ingestion, and national defense exercises also impact turtles in the Southwest Pacific DPS, although some of these threats have been minimized in recent years due to a variety of legislative actions. The magnitude of mortality from boat strikes may be in the high tens to low hundreds per year in Queensland (Limpus, 2009). This threat affects juvenile and adult turtles and may increase with increasing high-speed boat traffic in coastal waters. The magnitude of mortality from port dredging in Queensland may be in the order of tens of turtles or less per year (Limpus, 2009). A code of practice for port dredging operations was established in Queensland during the late 1990’s so this threat may be somewhat abated. National defense exercises have impacted green turtles and their habitat. Fairfax Island (and surrounding area) in the sGBR has been used for bombing practice. Population impacts have not been quantified, but because Fairfax Island is now part of the Capricornia Cays National Park (Limpus, 2009) this threat appears to be primarily mitigated.
**Toxic Compounds and Debris**

Toxic compounds and bioaccumulative chemicals threaten green turtles in the Southwest Pacific DPS. Poor health conditions (debilitation and death) have been reported in the southern Gulf of Carpentaria for green turtles, many of which had unusual black fat (Limpus, 2009 citing Kwan and Bell, 2003; Chapman, 2003; personal communications within Limpus, 2009). While the area is strictly outside of this DPS, Southwest Pacific DPS turtles may forage in the southern Gulf of Carpentaria and become vulnerable. The cause is not known, but it may be related to ecological effects of local flooding or the zinc-lead trade (Limpus, 2009 citing Kwan and Bell, 2003). The toxic compounds polychlorinated dibenzo-p-dioxins and dioxinsfurans (PCDD/Fs) have been found in green turtles in eastern Australia (Hermanussen et al., 2008; Limpus, 2009 citing Gaus et al., 2001), but the health impact has not been quantified. Heavy metal concentrations have also been reported in Australia (Gordon et al., 1998; Limpus, 2009 citing Dight and Gladstone, 1994 and Reiner, 1994), but again, the health impact has not been quantified. The magnitude of mortality from ingestion of synthetic material in Queensland is expected to be at least tens of turtles annually (Limpus, 2009).

**Climate Change and Natural Disasters**

Similar to other areas of the world, climate change and sea level rise have the potential to impact green turtles in the Southwest Pacific DPS. Green turtle populations could be threatened due to climate change effects on nesting grounds (Fuentes et al., 2011) as well as in marine habitats (Hawkes et al., 2009, Hamann et al. 2007). Potential effects of climate change include changes in nest site selection, skewed primary sex ratios, range shifts, and diets shifts (Hawkes et al., 2009).

Climate change and sea level rise have the potential to affect green turtles in the South Pacific DPS, yet a comprehensive assessment of these impacts is not available. Natural environmental events, such as cyclones or hurricanes, may affect green turtles in the Southwest Pacific DPS. These types of events may disrupt green turtle nesting activity, albeit on a temporary scale. It is reasonable to expect that climate change will result in future ecological changes for this DPS because relationships between climatic forces (such as the Southern Oscillation Index) and turtle reproduction have been documented (Limpus and Nicholls, 1988). In a study of the nGBR nesting assemblages, none were found to pass a threshold for being vulnerable to cyclonic activity; two were vulnerable to sea-level rise, and almost all sites in the study were expected to be vulnerable to increased temperatures by 2070 (Fuentes et al., 2011). Similar data is not available for other nesting sites.

Barnett and Adger (2003) identified projected increases in sea surface temperature, and not sea level rise, as the greatest long-term risk of climate change to atoll morphology. The Southwest Pacific DPS contains some atolls as well as coral reef areas which share some ecological characteristics with atolls. Barnett and Adger (2003) state that coral reefs, which are essential to the formation and maintenance of the islets located around the rim of an atoll, are highly sensitive to sudden changes in sea-surface temperature. Thus, climate change impacts could have long-term impacts on green ecology in the Southwest Pacific DPS, but it is not possible to project the impacts at this point in time.
12.2.6. Summary of Existing Conservation Efforts

Most countries in the Southwestern Pacific DPS have developed regional or national legislation to protect sea turtles and nesting habitats. National protective legislation generally regulates intentional killing, possession, and trade (Limpus, 2009; Maison et al., 2010).

The majority of nesting beaches (and often the associated internesting habitat) is protected in Australia, which is the country with the vast majority of the known nesting. For example, there is an ecotourism resort on Heron Island, but about 75 percent of the nesting activity occurs within protected areas (Limpus, 2009).

In Australia, the conservation of green turtles is governed by a variety of national and territorial legislation. Conservation began with 1932 harvest restrictions on turtles and eggs in Queensland in October and November, south of 17°S, and by 1968 the restriction extended all year long for all of Queensland (Limpus, 2009). As described in the preceding section, other conservation efforts have resulted in sweeping take prohibitions, implementation of bycatch reduction devises, improvement of shark control devises, and safer dredging practices, and the development of community based management plans with Indigenous groups. Australia has undertaken extensive marine spatial planning to protect nesting turtles and internesting habitat surrounding important nesting sites. The GBR’s listing on the United Nations Educational, Scientific and Cultural Organization’s World Heritage List in 1981 has increased the protection of habitats within the GBR World Heritage Area (Dryden et al., 2008).

In New Caledonia, 1985 fishery regulations contained some regional sea turtle conservation measures, and these were expanded in 2008 to include the EEZ, the Main Island, and remote islands (Maison et al., 2010). In Vanuatu, new fisheries regulations in 2009 prohibit the take, harm, capture, disturbance, possession, sale, purchase of or interference, import, or export of green turtles (Maison et al., 2010).

12.2.6.1. National Legislation and Protection

In addition to the international mechanisms, Australia, New Caledonia, and Vanuatu have developed legislation to protect sea turtles and/or nesting habitats. They are summarized below. However, the overall effectiveness of this country specific legislation is unknown at this time.

Australia

Green turtles in Australia are protected by the Environment Protection and Biodiversity Conservation Act of 1999 (http://www.environment.gov.au/epbc/) in addition to other conservation mechanisms. This Act is Australia’s main environmental legislation. It provides a legal framework to protect matters of environmental significance, including conserving Australian biodiversity. The Act lists green turtles as Vulnerable, migratory marine species (http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=1765) and provides recovery plans (Marine Species Section Approvals and Wildlife Division 2003) to reduce mortality, monitor the population, manage successful nesting, and protect habitats.
In addition to the national environmental legislation there is protection from territorial legislation (summarized from Limpus, 2009). The Great Barrier Reef Marine Park Act 1975 and Great Barrier Reef Marine Park Regulations 1983 lists green turtles as Protected. Threatened Species Protection Act 1995 in Tasmania lists green turtles as Vulnerable. Threatened Species Conservation Act 1995 in New South Wales lists green turtles as Vulnerable. Queensland’s Nature Conservation Act of 1992 lists green turtles as Vulnerable; Fisheries Act since 1968 lists green turtles as a protected species; the Wildlife Conservation Act 1950 of Western Australia lists green turtles as fauna that are rare or likely to become extinct; and the National Parks and Wildlife Act 1972 of South Australia lists green turtles as Vulnerable. Although not all of this legislation protects Southwest Pacific DPS turtles while they nest, it may afford protection while on forage grounds.

New Caledonia

Sea turtle conservation measures vary within New Caledonia (as summarized by Maison et al., 2010). The take of sea turtles and their eggs is prohibited from November 1 through March 31 in the Loyalty Islands province. It is not permitted to capture, sell, purchase, or disturb any marine turtle species or nest at anytime in the EEZ, the Main Island (Northern and Southern provinces), and remote islands. The use of sea turtle handling equipment (de-hooker, line-cutter, etc.) is required commercial fisheries, and regulations generally prohibit the export or import of marine turtles or turtle parts or products.

Vanuatu

Within Vanuatu, the Vanuatu Fisheries Act of 2009 (as summarized by Maison et al., 2010) prohibits the take, harm, capture, disturbance, possession, sale, purchase of or interference with turtle nests, and the import, or export of green, turtles or their products.

12.2.6.2. International Instruments

Several treaties and/or regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the Southwest Pacific DPS (Maison et al., 2010). The international instruments listed below apply to sea turtles found in the Southwest Pacific DPS and are described in Appendix 5.

- Convention on Biological Diversity
- Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention)
- Convention on the Conservation of Migratory Species of Wild Animals
- Convention on International Trade in Endangered Species of Wild Fauna and Flora
- Food and Agricultural Organization Technical Consultation on Sea Turtle-Fishery Interactions
- Forum Fisheries Authority
- Indian Ocean Tuna Commission
- Indian Ocean South-East Asian Sea Turtle Memorandum of Understanding
- Inter-American Tropical Tuna Commission
12.3. Assessment of Significant Portion of its Range (SPR)

The SRT reviewed the information on threats and extinction risk within each DPS to determine whether there were portions of the range of the DPS that might be considered “significant portions of the range” in accordance with the definition of endangered and threatened species and the draft policy that interprets that term (see Section 3.4).

The nesting abundance in this DPS is high and distributed throughout the region, but with higher reported abundances in the Great Barrier Reef in the western part of the DPS. The SRT noted the low Raine Island nest productivity and conservation measures related to increasing hatching success. Although there are some regional differences in threats, on the whole, the negative effect of threats are thought to be fairly uniform, particularly in the western part of the DPS where the bulk of the population nests. The SRT concluded that the need to consider a significant portion of the range does not apply to this DPS.

12.4. Assessment of Extinction Risk

For the SRT’s assessment of extinction risk for green turtles in the Southwest Pacific DPS, there were two separate sets of ranking exercises: One focusing on the importance that each SRT member placed on each of the six critical assessment elements for this region (Table 12.3), and a second which reflects the SRT members’ expert opinion about the probability that green turtles would fall into any one of various extinction probability ranges (Table 12.4; see Section 3.3 for discussion of this process).
Table 12.3. Summary of ranks that reflect the importance placed by each SRT member on the critical assessment elements considered for the Southwest Pacific DPS. See Section 3.3 for details on the six elements and the voting process. For Elements 1-4, higher ranks indicate higher risk factors.

<table>
<thead>
<tr>
<th>Critical Assessment Elements</th>
<th>Element 1</th>
<th>Element 2</th>
<th>Element 3</th>
<th>Element 4</th>
<th>Element 5</th>
<th>Element 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abundance (1 to 5)</td>
<td>1.17</td>
<td>1.67</td>
<td>1.50</td>
<td>1.42</td>
<td>-0.67</td>
<td>0.58</td>
</tr>
<tr>
<td>Trends / Productivity (1 to 5)</td>
<td>0.17</td>
<td>0.19</td>
<td>0.26</td>
<td>0.19</td>
<td>0.19</td>
<td>0.23</td>
</tr>
<tr>
<td>Spatial Structure (1 to 5)</td>
<td>1–3</td>
<td>1–3</td>
<td>1–4</td>
<td>1–3</td>
<td>(-2)–0</td>
<td>0–2</td>
</tr>
<tr>
<td>Diversity / Resilience (1 to 5)</td>
<td>5–Factor Analyses (-2 to 0)</td>
<td>72.50</td>
<td>9.08</td>
<td>10.58</td>
<td>5.42</td>
<td>2.42</td>
</tr>
<tr>
<td>Conservation Efforts (0 to 2)</td>
<td>10.74</td>
<td>2.27</td>
<td>5.20</td>
<td>3.39</td>
<td>1.43</td>
<td>0.00</td>
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<td>SEM</td>
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<td>0</td>
<td>0</td>
<td>0</td>
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<tr>
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</tr>
</tbody>
</table>

With respect to the importance of rankings for the six critical assessment elements, the average of the scores for the first four elements (Abundance, Productivity, Spatial Structure, and Diversity) was similar and had relatively low values, ranging from 1.17 to 1.67.

SRT members also generally thought that future threats not yet reflected in nester abundance or not yet experienced by the population weighed slightly heavier in their risk assessment voting (average of 0.67) than did any conservation efforts that may emerge in the future (average of 0.58). SRT members had diverse opinions when considering the six critical assessment elements. With respect to the diversity of opinions among the SRT members when considering these elements, the largest range in rankings (i.e. voter opinion) was noted for Spatial Structure (w/ ranks from 1 to 4).

Table 12.4. Summary of Green Turtle SRT member expert opinion about the probability that the Southwest Pacific DPS will reach a critical risk threshold within 100 years. Each SRT member assigned 100 points across the rank categories. The continuum in Row 1 has categories with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability Of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1-5%</th>
<th>6-10%</th>
<th>11-20%</th>
<th>21-50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN ASSIGNED POINTS</td>
<td>72.50</td>
<td>9.08</td>
<td>10.58</td>
<td>5.42</td>
<td>2.42</td>
<td>0.00</td>
</tr>
<tr>
<td>SEM</td>
<td>10.74</td>
<td>2.27</td>
<td>5.20</td>
<td>3.39</td>
<td>1.43</td>
<td>0.00</td>
</tr>
<tr>
<td>Min</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Max</td>
<td>99</td>
<td>21</td>
<td>60</td>
<td>40</td>
<td>15</td>
<td>0</td>
</tr>
</tbody>
</table>

Of the critical risk threshold categories describing the probability that the Southwest Pacific DPS will reach a critical risk threshold within 100 years, SRT member voted overwhelmingly in the

Comment [A54]: Could this be due to Raine Island?

Response: it could be due to Raine Island, but the level of detail in the written comments don’t support a more detailed description about spatial structure and Raine. Certainly folks did have habitat concerns about Raine.
‘<1%’ risk range (mean of 72.5). The ‘1–5%’ and ‘6–10%’ categories had much lower average points (mean of 9.08 and 10.58, respectively). The scores decreased by about half for the ‘11–20%’ category (mean of 5.42) and decreased by half again for the ‘21–50%’ category (mean of 2.42). The ‘>50%’ category received no points by any SRT member. The range of scores in the individual risk categories was quite high, particularly for the ‘<1%’ category which ranged from 0 to 99.

In their vote justifications for this DPS, most members cited high nester abundance, including two large nesting sites one of which is among the world’s largest. Additional factors that were cited included the positives trends, PVA results, robust spatial structure and diversity (noting dispersed nesting on continent and islands), ancestral haplotypes, different ecological types of foraging areas, and nesting throughout year. Many members also included comments about substantial conservation measures and relatively well-managed threats. Over half of the vote justifications included concerns about future risks, including concerns about climate change and Raine Island productivity. Concerns about climate change included temperature changes, effects on the GBR habitat, increased tropical storms, and loss of nesting beaches due to erosion and sea level rise. Concerns about Raine Island overlapped somewhat with the climate change concerns and included the condition of nesting habitat and low productivity related to nesting conditions.

Figure 12.4. Bar graph depicting Green Turtle SRT member expert opinion about the probability that the East Pacific DPS will reach a critical risk threshold within 100 years.

12.5. Synthesis and Integration
During the analysis of the Southwest Pacific DPS’s status, an integrated approach was taken by the SRT to consider the many critical elements described earlier. The Southwest Pacific DPS is characterized by relatively high levels of green turtle nesting abundance and contains the GBR, the largest coral reef system in the world, as well as continental coastline, islands, and atolls. Although individuals from this DPS may share common foraging grounds with other DPSs, this mixing does not appear to apply to nesting sites.

The Southwest Pacific DPS has a nesting female abundance greater than 80,000 among 12 broadly defined nesting locations. The trends in nester abundance at the two index beaches (Raine Island and Heron Island) are stable or increasing. The spatial structure of this DPS extends over a large geographic area, with several large nesting sites through the geographic range of this DPS, and includes both continental and insular nesting. This region has high genetic diversity resulting from a mix of highly divergent lineages, some of which are among the oldest lineages found in \textit{C. mydas}. There were concerns about climate change in general and the nesting habitat at Raine Island in particular. On average, the SRT thought that these four elements (abundance, trends, spatial structure, and diversity) represented either low or very low risk to the viability of the DPS.

Many of the threats to this DPS are at least partially mitigated by conservation measures. In the sGBR threats are well managed, harvest is low, population increasing; however, in the nGBR there are concerns for Raine Is, but there are many other smaller rookeries for which nesting success is normal. Harvest is higher in the nGBR but has been well managed in last 3 years with community based management. In the Coral sea there are few known threats and it is remote and well managed from human threats.

The threats to this Southwest Pacific DPS include historic commercial harvest, contemporary directed harvest, incidental bycatch, shark control programs, boat strikes, port dredging, debris, national defense, disease, predation, toxic compounds, and climate change. Conservation efforts have resulted in sweeping take prohibitions, implementation of bycatch reduction devices, improvement of shark control devices, and safer dredging practices. Australia, in particular, has undertaken extensive marine spatial planning to protect nesting turtles and internesting habitat surrounding important nesting sites. On average, the SRT thought that threats considered in the 5-factor / threats analysis were likely to have minimal to moderate effects beyond what was reflected in the Abundance, Trends, Spatial Structure, and Diversity / Resilience elements and, on average, the SRT thought that conservation measures were likely to have minimal to moderate effects beyond what was reflected in the these elements.

The SRT determined the likelihood of reaching a critical risk threshold of extinction within 100 years was relatively low (72.5 percent of votes cast for the ‘<1%’ likelihood category), there was variation among SRT members with some members indicating far greater risks. These results reflect the view that, while the DPS shows strength in many of the critical elements, there are still concerns about future risks including climate change and habitat degradation. It could also reflect differences in how people weighted their scores in terms of placing more/less importance to the sGBR or nGBR. Recall that one half of the DPS is well managed and increasing and the other half of the DPS is stable but warrants concern.

Comment [A55]: It could also reflect differences in how people weighted their scores in terms of placing more/less importance to the sGBR or nGBR. One half of the DPS is well managed and increasing and the other half of the DPS is stable but warrants concern.

Response: We agree and have added these possibilities to the text.
13. CENTRAL SOUTH PACIFIC DPS (DPS #9)

13.1. DPS Range and Nesting Distribution

The Central South Pacific DPS extends north from northern New Zealand to Fiji, Tuvalu, and Kiribati and east to include French Polynesia. Its open ocean polygonal boundary endpoints are (clockwise from the northwest-most extent): 9°S, 185°E W to 9°S, 235°E W to 40°S, 264°E W to 40°S, 176°E, to 13°S, 171°E, and back to the 9°S, 185°E W northwest extent. This DPS includes a longitudinal expanse of 7,500 km—from Easter Island, Chile in the east to Fiji in the west, and encompasses American Samoa, French Polynesia, Cook Islands, Fiji, Kiribati, Tokelau, Tonga, and Tuvalu. Nesting occurs sporadically throughout the geographic distribution of the population, with isolated locations having relatively low to moderate nesting activity (Figure 13.1).

![Figure 13.1. Nesting distribution of green turtles in the Central South Pacific DPS.](image)

Size of circles indicates estimated nester abundance (see Section 13.2.1). Locations marked with ‘×’ indicate nesting sites lacking abundance information.

Green turtles departing nesting grounds in this DPS travel throughout the South Pacific Ocean. Post-nesting green turtles tagged in the early 1990s from Rose Atoll returned to foraging grounds in Fiji and French Polynesia (Craig et al., 2004). Nesters tagged in French Polynesia migrated west after nesting to various sites in the western South Pacific (Tuato’o-Bartley et al., 1993).
addition to nesting beaches, green turtles are found in coastal waters (White, 2013; White and Galbraith, 2013), but in-water information in this population is particularly limited.

13.2. **Critical Assessment Elements**

In the evaluation of extinction risk for green turtles in the Central South Pacific DPS, the SRT considered six distinct elements, each of which are discussed below: (1) Nesting Abundance, (2) Population Trends (3) Spatial Structure, (4) Diversity / Resilience, (5) Five-Factor Threat Analyses, and (6) Existing Conservation Efforts.

13.2.1. **Nesting Abundance**

Green turtle nesting in the Central South Pacific DPS is geographically widespread at low to moderate levels (Table 13.2). The most abundant nesting area is Scilly Atoll, French Polynesia, which in the early 1990s was estimated to host 300–400 nesters annually (Balazs et al., 1995), and we estimate having a total nester abundance of 1,050 breeding females (Table 13.1). The most recent information is for American Samoa, with the majority of nesting at Rose Atoll and sporadic nesting on Tutuila and Swains Islands; sub-adult and adult turtles occur in low abundance in nearshore waters around Tutuila, Ofu, Olosega, Ta’u, and Swains islands (NMFS and USFWS, 1998; Maison et al., 2010). Historically, 100–500 females nested annually at Canton Island, Kiribati (Mast, 2011). Historical baseline nesting information in general is not widely available in this region, but exploitation and trade of green turtles throughout the region is well-known (Groombridge and Luxmoore, 1989). No long-term monitoring programs are currently available at beaches in this population.

Based on available data, we estimate there are nearly 3,000 nesters in this DPS. However, the largest nesting site, Scilly Atoll, which comprises roughly one third of the entire nesting abundance, was last monitored in the early 1990s (Balazs et al., 1995) and has reportedly significantly declined in the past 30 years as a result of commercial exploitation (Petit, 2013). No sites have long-term monitoring programs, and no single site has had standardized surveys for even 5 contiguous years. Most nesting areas are in remote, low-lying atolls that are logistically difficult to access. Unsurprisingly, many nesting areas (21 of 57, or 37 percent) only have qualitative information that nesting is present, indicating that there is still much to learn about green turtle nesting in this region (Table 13.2). As these unquantified rookeries most likely each have a female abundance in the 1–100 range, their collective sum is probably less than 700 nesters. When added to our ca. 3,000 total, this DPS likely has ca. 3,600 nesters.
Table 13.1. Summary of green turtle nesting activity in the Central South Pacific DPS. Data are organized by country, nesting site, monitoring period, and estimated total nester abundance [(Total Counted Females/Years of Monitoring) x Remigration Interval], and represents only those sites for which there were sufficient data to estimate number of females. Many nesting sites in the Central South Pacific DPS are data deficient and estimates could not be made for those beaches. For a list of references for these data, see Appendix 2.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YEARS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Am. Samoa (USA)</td>
<td>Rose Atoll</td>
<td>2006–2012</td>
<td>105</td>
</tr>
<tr>
<td>Am. Samoa (USA)</td>
<td>Swains Atoll</td>
<td>2007–2013</td>
<td>23</td>
</tr>
<tr>
<td>Am. Samoa (USA)</td>
<td>Tutuila</td>
<td>2007–2013</td>
<td>3</td>
</tr>
<tr>
<td>Cook Islands</td>
<td>Palmerston Atoll</td>
<td>2010</td>
<td>149</td>
</tr>
<tr>
<td>Cook Islands</td>
<td>Tongareva Atoll</td>
<td>2011-2013</td>
<td>184172</td>
</tr>
<tr>
<td>Cook Islands</td>
<td>Rakahanga</td>
<td>2009-2012</td>
<td>20</td>
</tr>
<tr>
<td>Fiji</td>
<td>Nanuku Levu</td>
<td>2006</td>
<td>96</td>
</tr>
<tr>
<td>Fiji</td>
<td>Nukumbalati</td>
<td>2006</td>
<td>96</td>
</tr>
<tr>
<td>French Polynesia</td>
<td>Scilly Atoll</td>
<td>1991</td>
<td>1,050</td>
</tr>
<tr>
<td>French Polynesia</td>
<td>Tuamotus</td>
<td>1930</td>
<td>225</td>
</tr>
<tr>
<td>French Polynesia</td>
<td>Mopelia</td>
<td>2010</td>
<td>168</td>
</tr>
<tr>
<td>French Polynesia</td>
<td>Motu One</td>
<td>1991</td>
<td>99</td>
</tr>
<tr>
<td>French Polynesia</td>
<td>Bora Bora</td>
<td>2010</td>
<td>33</td>
</tr>
<tr>
<td>French Polynesia</td>
<td>Tetiaroa</td>
<td>2010</td>
<td>25</td>
</tr>
<tr>
<td>French Polynesia</td>
<td>Tikehau</td>
<td>2007–2010</td>
<td>11</td>
</tr>
<tr>
<td>French Polynesia</td>
<td>Tupai</td>
<td>1995</td>
<td>6</td>
</tr>
<tr>
<td>French Polynesia</td>
<td>Maupiti</td>
<td>2010</td>
<td>3</td>
</tr>
<tr>
<td>French Polynesia</td>
<td>Maiao</td>
<td>2009</td>
<td>3</td>
</tr>
<tr>
<td>Kiribati</td>
<td>Enderbury</td>
<td>2002</td>
<td>129</td>
</tr>
<tr>
<td>Kiribati</td>
<td>Nikumaroro</td>
<td>2002</td>
<td>56</td>
</tr>
<tr>
<td>Kiribati</td>
<td>O [Kanton]</td>
<td>2002</td>
<td></td>
</tr>
<tr>
<td>Kiribati</td>
<td>Manra</td>
<td>2002</td>
<td>24</td>
</tr>
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<td>Tarawa</td>
<td>2007</td>
<td>17</td>
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<tr>
<td>Kiribati</td>
<td>Teraina (Washington)</td>
<td>1990s</td>
<td>15</td>
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<tr>
<td>Kiribati</td>
<td>Malden</td>
<td>1990s</td>
<td>15</td>
</tr>
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<td>Kiribati</td>
<td>Phoenix</td>
<td>2002</td>
<td>9</td>
</tr>
<tr>
<td>Kiribati</td>
<td>Orona</td>
<td>2002</td>
<td>8</td>
</tr>
<tr>
<td>Kiribati</td>
<td>Caroline</td>
<td>1990s</td>
<td>8</td>
</tr>
<tr>
<td>Kiribati</td>
<td>McKean</td>
<td>2000</td>
<td>6</td>
</tr>
<tr>
<td>Kiribati</td>
<td>Birnie</td>
<td>2002</td>
<td>5</td>
</tr>
</tbody>
</table>
### Table 13.2. The distribution of green turtle nester abundance in the Central South Pacific.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YEARS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Tokelau</td>
<td>Nukunonu</td>
<td>1970s</td>
<td>210</td>
</tr>
<tr>
<td>Tokelau</td>
<td>Fakaofu</td>
<td>1970s</td>
<td>90</td>
</tr>
<tr>
<td>Tokelau</td>
<td>Atafu</td>
<td>1970s</td>
<td>60</td>
</tr>
<tr>
<td>Tonga</td>
<td>Fonuaika</td>
<td>2007</td>
<td>11</td>
</tr>
<tr>
<td>Tonga</td>
<td>Nukulei</td>
<td>2007</td>
<td>5</td>
</tr>
<tr>
<td>Tonga</td>
<td>Luanamo</td>
<td>2007</td>
<td>2</td>
</tr>
<tr>
<td>Tuvalu</td>
<td>Funafuti</td>
<td>2006</td>
<td>90</td>
</tr>
<tr>
<td>UK Overseas Territory</td>
<td>Henderson</td>
<td>1991</td>
<td>30</td>
</tr>
</tbody>
</table>

### Table 13.2.1. Nester Abundance Distribution in the DPS 9

<table>
<thead>
<tr>
<th>NESTER ABUNDANCE</th>
<th># NESTING SITES</th>
</tr>
</thead>
<tbody>
<tr>
<td>n/a</td>
<td>21</td>
</tr>
<tr>
<td>1–10</td>
<td>11</td>
</tr>
<tr>
<td>11–50</td>
<td>11</td>
</tr>
<tr>
<td>51–100</td>
<td>7</td>
</tr>
<tr>
<td>101–500</td>
<td>6</td>
</tr>
<tr>
<td>501–1000</td>
<td>0</td>
</tr>
<tr>
<td>1001–5000</td>
<td>1</td>
</tr>
<tr>
<td>5001–10000</td>
<td>0</td>
</tr>
<tr>
<td>10001–100000</td>
<td>0</td>
</tr>
<tr>
<td>&gt;100,000</td>
<td>0</td>
</tr>
<tr>
<td>TOTAL SITES</td>
<td>57</td>
</tr>
<tr>
<td>TOTAL ABUNDANCE</td>
<td>2,914,902</td>
</tr>
<tr>
<td>% at LARGEST NESTING SITE</td>
<td>36%</td>
</tr>
</tbody>
</table>

13.2.2. Population Trends

Green turtle temporal population trends in the Central South Pacific DPS are poorly understood, with not even a single nesting site having five contiguous years of standardized monitoring that span entire nesting seasons. Therefore, we have no data to conduct a PVA, or even a simple bar chart of annual nesting activity. Partial and inconsistent monitoring from the largest nesting site in this DPS, Scilly Atoll, suggests significant nesting declines from persistent and illegal commercial harvesting (Petit, 2013). Nesting abundance is reported to be stable to increasing at Rose Atoll, Swains Atoll, Tetiaroa, Tikihau, and Maiao. However, these sites are of moderate to low abundance and in sum represent less than 16 percent of the population abundance at Scilly Atoll alone (Table 13.1). Nesting abundance is reported to be stable to increasing at Tongareva
Atoll (White and Galbraith, 2013). The uncertainty surrounding these above trends, and the general dearth of long-term monitoring and data from this DPS, presents significant challenges to any formal quantitative trend analyses. For a list of references on trend data, see Appendix 3.

13.2.3. Spatial Structure

When examining spatial structure for the Central South Pacific DPS, the SRT examined three lines of evidence, including genetic data, flipper and satellite tagging, and demographic data.

Genetic sampling in the Central South Pacific DPS has been limited and many of the small isolated nesting sites that characterize this region have not been covered. Genetic sampling is currently underway at Tongareva Atoll, Cook Islands (M. White, unpubl. data). Based on limited sampling, there is evidence of significant spatial structuring. Within the DPS, there is significant population substructuring (pairwise $F_{ST}$ $0.53$, $p<0.005$) between American Samoa and French Polynesia (P. Dutton, NMFS, unpubl. data). The samples from American Samoa were collected across four locations (Swains Island, Tutuila, Ofu and Rose Atoll) that had both low sample sizes ($n = 1–8$) and were a great distance from each other (160–500 km). However, these were pooled to represent American Samoa as they shared haplotypes and were significantly distinct from French Polynesia, represented by one sampled nesting site ($n = 9$) at Mopelia (P. Dutton, NMFS, unpubl. data).

Flipper tag returns and satellite tracking studies demonstrate that post-nesting females travel the complete geographic breadth of this DPS, from French Polynesia in the east to Fiji in the west, and sometimes even slightly beyond (Tuato‘o-Bartley et al., 1993; Craig et al., 2004; Maison et al., 2010; White, 2012) and the Philippines (Trevor, 2009). The complete extent of migratory movements is unknown.

Demographic studies of green turtles do not reveal any structuring of traits within the DPS. Limited demographic information is available for green turtles in the Central South Pacific DPS. Nesters at Scilly Atoll, French Polynesia in one study of five females averaged 95.6 cm CCL (Hirth, 1980) and in another study of 51 females averaged 103 cm SCL (Balazs et al., 1995). Nesters at Rose Atoll, American Samoa averaged 94.7 cm CCL ($n=68$, K. Van Houtan, NMFS, unpubl. data, 2013). Five nesters in Tokelau ranged from 102–104 cm CCL (Balazs, 1983) and had a 14-day interval between clutches. Peak nesting occurs from August to November at Rose Atoll (Craig et al., 2004), occurs in November in American Samoa (Tuato‘o-Bartley et al., 1993), occurs in January to February at Pitcairn Island (Brooke, 1995), and occurs from June to December in Tokelau (Balazs, 1983). Demographic information from nest-level inventories is not available for this population. Typically studied population variables such as mean nesting size, nesting season, inter-nesting interval, clutch size, hatching success, nesting season, and clutch frequency suggest a low level of population structuring of green turtles within this DPS (Tuato‘o-Bartley et al., 1993; Craig et al., 2004; White, 2012; White and Galbraith, 2013).

13.2.4. Diversity and Resilience

The parameters considered under this critical element include the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation (e.g. high vs. low
beach face, insular vs. continental nesting sites), and the genetic diversity within the DPS. Parameters such as these are important considerations for assessing the potential impact of catastrophic events such as storms, sea level rise, and disease.

The Central South Pacific has a broad geographical area, but the nesting sites themselves exhibit little diversity. Most nesting sites are located in low-lying coral atolls or oceanic islands as the region has no true continental land mass. Local nesting density is sparse spatially, typically spread over > 10 km stretches of beach and is also low in terms of abundance (Table 13.2). Only one nesting site (Scilly Atoll with 1,050 females) has a nester abundance exceeding 250. Foraging areas are mostly coral reef ecosystems, with seagrass beds in Tonga and Fiji being a notable exception.

In summary, most nesting sites in this DPS are in remote low-lying atolls, have low abundance, and nesting is at low spatial densities. Mitochondrial DNA studies based on very limited sampling indicates there are at least two genetic stocks in the Central South Pacific DPS, with a moderate level of diversity and presence of unique haplotypes (P. Dutton, NMFS, unpubl. data).

13.2.5. Analysis of Factors Listed Under ESA Section 4(a)(1)

Section 4(a)(1) of the ESA lists five factors (threats) that must be considered when determining the status of a species. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

The following information on these factors /threats pertains to green turtles found in the Central South Pacific DPS, although it should be noted that there is limited information for certain threats in this DPS. Because foraging green turtles or migratory routes of green turtles from this DPS also are found within the boundaries of the Central West Pacific, the Southwest Pacific, and East Indian-West Pacific DPSs, the narrative for those DPSs should also be consulted.

13.2.5.1. Factor A: Destruction or Modification of Habitat or Range

The Central South Pacific DPS of the green turtle is negatively affected by ongoing changes in both its terrestrial and marine habitats as a result of land and water use practices. While threats to these habitats exist and are a potential threat to this DPS, the exact magnitude is difficult to determine given limited information.

Terrestrial Zone

Nesting in the Central South Pacific DPS is geographically widespread with the majority of nesting sites being remote and not easily accessed, and at low-lying oceanic islands or coral
atolls. The largest nesting site for this DPS is at Scilly Atoll in French Polynesia. A summary of available information regarding threats relevant to nesting areas in this DPS is provided below.

Balazs et al. (1995) report that the earliest human settlement (for copra production) at Scilly Atoll in French Polynesia appears to have occurred around 1952. It is unclear how much of an effect human habitation of the atoll has had, or is having, on this nesting site.

The major nesting site for green turtles in American Samoa is at Rose Atoll, with sporadic nesting on Tutuila and Swains Islands although Amerson et al. (1982a, 1982b) reports that green turtles historically nested in greater numbers on beaches of islands other than Rose Atoll. Because Rose Atoll is uninhabited, there are currently no threats related to development or other problems associated with human villages or settlements (e.g., beach construction, beach armoring, beachfront lighting, removal of native vegetation), nor does any sand mining occur there. Sand mining does occur on other inhabited islands in American Samoa, but is not thought to be a significant threat (NMFS and FWS, 1998). American Samoa is considering the authorization of archeological monitoring or excavation activities on the beaches of Rose Atoll that could affect green turtle nesting and nests although these effects, due to foot traffic (sand compaction) over nests and excavations, may be minimal (FWS, 2012).

In the populated islands of American Samoa, such as Tutuila, continuous incremental loss of habitat has occurred due to varied activities of human populations (Tuato’o-Bartley et al., 1993; NMFS and FWS, 1998). Indeed, human population growth and attendant village expansion and development on Tutuila Island has resulted in decreasing usage of some Tutuila beaches by nesting turtles and pre-emption of some green turtle nesting beaches (Tuato’o-Bartley et al., 1993). For example, the complete removal of nesting habitat to make way for structures such as seawalls impacts nesting areas in more developed areas within this DPS (Saili, 2005). Turtles on Tutuila, possibly disoriented by land based lights, are subject to mortality from cars (A. Tagarino, American Samoa DMWR, pers. comm., 2013). Lighting is a potential problem affecting the quality of the nesting habitat on Ofu nesting beach as well (Tagarino, 2012).

In Samoa, degradation of habitat through coastal development and natural disasters as cited in SPREP (2012) remains a threat (J. Ward, Ministry of Natural Resources and Environment, Samoa, pers. comm., 2013).

In Kiribati, historical destruction (bulldozing) of the vegetation zone next to nesting beach on Canton Island in the Phoenix Islands occurred during World War II and may have negatively affected the availability of a portion of nesting beach area (Balazs, 1975). The remoteness of these Islands and minimal amount of study of sea turtles of this area makes recent information on nesting beach condition and threats difficult to obtain.

Nesting also occurs in Tonga, Tuvalu, and Fiji. Coastal erosion in Tonga is reported as a major problem for turtle nesting (Bell et al., 2010). In Tuvalu, coastal erosion on known sea turtle nesting sites and loss of coastal vegetation are identified threats (Alefaio and Alefaio, 2006). Weaver (1996) notes that sea turtles are negatively affected in Fiji by modification of nesting beaches.
The major nesting site for green turtles in the Cook Islands is at Tongareva Atoll, with one motu (cay) Mangarongaro being the paramount nesting beach nationally (White 2012). Because Mangarongaro motu is uninhabited, and only accessible by small boat, there are likely no threats related to development or other problems associated with human villages or settlements (e.g., beach construction, beach armoring, beachfront lighting, removal of native vegetation), nor does any sand mining occur there.

Elsewhere in the Cook Islands, sand extraction (for building purposes) and building developments are reported as potential threats to sea turtles; for instance, the best potential site at Tauhunu motu on Manihiki appears to be no longer used for nesting (White, 2012). Bradshaw and Bradshaw (2012) mention sand extraction at Mauke; however, it does not appear to have a large impact at this particular site. Light pollution is a recognized threat in the Cook Islands (White, 2012, 2013).

No information on threats to green turtle nesting habitat could be found for the Territory of the Wallis and Futuna Islands.

Like other atoll and island areas, climate change and sea level rise are a threat to the areas throughout this DPS. Climate change has been recognized as a potential threat to sea turtles, including terrestrial habitat of the Central South Pacific. Climate change is discussed further below in 6.2.9.5.

**Neritic/Oceanic Zone**

Little is known regarding the status of the foraging habitat and threats found in French Polynesia. (Balazs et al., 1995).

NMFS and FWS (1998) noted that degradation of coral reef habitats on the south side of Tutuila Island, American Samoa is occurring due to sedimentation from erosion on agricultural slopes and natural disasters. Ship groundings are also potential threats to habitat in American Samoa. For example, a ship grounded at Rose Atoll in 1993, damaging reef habitat and spilling 100,000 gallons of fuel and other contaminants (FWS, 2012). In the nearby neighboring country of Samoa, coastal and marine areas have been negatively impacted by pollution (Government of Samoa, 1998).

Fiji appears to be an important foraging area for green turtles of this DPS. Sea turtles have been negatively affected by alteration and degradation of foraging habitat and to some extent pollution or degradation of nearshore ecosystems (Batibasaga et al., 2006). Jit (2007) also suggests that sea turtles in Fiji are threatened by degradation of reefs and seagrass beds. Given that turtles outside of Fiji appear to use this foraging habitat, negative effects to this foraging area have important implications for the entire DPS. Tourism development on the eastern coast of Viti Levu could negatively impact sea turtle foraging sites (Jit, 2007).

While no sea turtle habitat specific studies have been conducted in the Phoenix Islands (Kiribati), it has been suggested that they are a healthy example of central Pacific atoll coral reef communities given their exclusion from long term human impacts (Stone et al., 2001).
In Tonga, in water habitat is being affected by anthropogenic activities. Heavy sedimentation and poor water quality have killed patch reefs; heavy sedimentation, high nutrients and high turbidity are negatively impacting seagrasses; and human activities are negatively impacting mangroves (Prescott et al., 2004).

Although Palmyra Atoll is now protected, it was altered by U.S. military activities during World War II through dredging, connection, and expansion of islets (Sterling et al., 2013). Proposed restoration activities could result in sediments and pollutants being released and negatively affecting feeding grounds and other habitats; however long term impacts could be beneficial (Sterling et al., 2013). Overall effects of these potential activities are currently unclear.

During In 2012 Rarotonga’s main harbor at Avatiu, Cook Islands, was dredged and extended. The adjacent reef was heavily impacted by sediment and the resident green turtles left; it is unclear how many of these individuals have returned (White pers. com).

Tongareva Atoll, Cook Islands, has juvenile and adult green turtles present throughout the year. The atoll is a mainly subsistence way of life and human impacts are thus low. In 2012 two algal blooms (green tides) were reported from the lagoon; their cause remains unknown (White, 2012).

No information on threats to green turtle neritic or oceanic habitat could be found for the Territory of the Wallis and Futuna Islands. Sea level rise as a result of climate change has been recognized as a potential threat to sea turtles, including neritic/oceanic habitat of the Central South Pacific, and is discussed further below in 6.2.9.5.

13.2.5.2. Factor B: Overutilization

Overutilization for commercial and subsistence purposes likely was a factor that contributed to the historical decline of this DPS. Despite national laws in various countries, legal and illegal harvest of green turtles and eggs for human consumption continues as a significant threat to this DPS

Egg and Turtle Harvest

At Scilly Atoll in French Polynesia local residents (approximately 20 to 40 people) are allowed to take 50 adults (both sexes) per year from a nesting population that could be as low as 300–400 (Balazs et al., 1995; Allen, 2007). Balazs et al. (1995) reported that declines in nesting green turtles at the important areas of Scilly, Motu-one, and Mopelia have occurred due to commercial exploitation for markets in Tahiti, as well as exploitation due to human habitation. Illegal harvest of sea turtles has been reported for French Polynesia by Te Honu Tea (2007). Brikke (2009) conducted a study on Bora Bora and Maupiti islands and reported that sea turtle meat remains in high demand and that fines are rarely imposed, authorities have poor control at nesting sites and landings are not adequately monitored, poaching is uncontrollable due to the fragmented, insular nature of French Polynesia, and there is poor enforcement of protective laws.
Human consumption has had a significant impact on green turtles in the Central South Pacific DPS. Hirth (1992) reports the exploitation of green turtles for eggs, meat, and parts has occurred throughout the South Pacific Region including, but not limited to, American Samoa, Cook Islands, Fiji Islands, French Polynesia, and Kiribati. Allen (2007) notes that in Remote Oceania (which includes this DPS) sea turtles were important in traditional societies but, despite this, have experienced severe declines since human colonization approximately 2,800 years ago. At western contact, some of the islands supported sizable human populations resulting in intense pressures on local coastal fisheries.

Directed take in the marine environment has been a significant source of mortality in American Samoa, and turtle populations have seriously declined (Tuato’o-Bartley et al., 1993; NMFS and FWS, 1998). Take of sea turtle eggs or sea turtles is illegal (the ESA applies to this territory). Grant et al. (1997) reported that even though there had been educational efforts relating to sea turtles, some turtles and eggs were still illegally taken. NMFS and FWS (1998) noted directed take as a significant source of mortality in American Samoa. The extent of current illegal take may be less than in the past (A. Tagarino, American Samoa DMWR, pers. comm., 2013); however, actual levels are unknown. Turtles from American Samoa migrate to other countries (e.g., Fiji, Samoa, French Polynesia) where turtle consumption is legal or occurs illegally. For example, there is a documented instance of two turtles which were tagged at Rose Atoll being captured and eaten in Fiji (Craig, 1993; Tuato’o-Bartley et al., 1993). This illustrates the complexity of threats affecting South Pacific green turtles, including green turtles of the Central South Pacific DPS. Animals are protected in some countries, but when they migrate to other countries they face the threat of harvest.

Turtles have been traditionally harvested for food and shells in the country of Samoa. Shells are used for hooks and jewelry including a headpiece used by a princess during important dance ceremonies (Craig, 1993). Witzell (1982) documented green turtles to have been taken by spear or by hand by skin divers or occasionally encircled with nets placed by fishermen. Turtles have historically been a valuable resource, often sold in Apia to affluent Samoans for important celebrations (Witzell, 1982), although current information is lacking on this practice. Over-exploitation of turtles has negatively affected local populations (Government of Samoa, 1998). Unsustainable harvest (direct take for meat) remains a major threat to green turtles in Samoa (J. Ward, Government of Samoa, pers. comm. 2013).

In Fiji, Weaver (1996) identified the contemporary harvest and consumption of turtles by humans for eggs, meat, and shells as a significant threat for sea turtles. Commercial harvest (a major threat, as well as subsistence and ceremonial harvest, are all contributing factors. Rupeni et al. (2002) report that green turtles are threatened from traditional harvesting for ceremonial purposes, as well as from subsistence and commercial harvesting for meat, eggs, and shell (turtles captured for general consumption and sale in local markets). Although a thorough assessment of these threats is not available, Batibasaga et al. (Batibasaga et al., 2006) note that sea turtles in Fiji have been substantially overfished since the 1980s, and report that an estimated 400–500 green turtles are killed in Fiji each year. Laveti and MacKay (2009) found that open sale of turtles in markets no longer occurs, but report that anecdotal information suggests substantial catch of turtles for subsistence, traditional use, and possible black market commercial sales. Their market research on the island of Viti Levu from April 2006 to 2007 found 29 green
turtle carapaces (average price of US $42). Illegal harvest of sea turtles by villages in Fiji for household consumption still occurs, and the rules that allow traditional take are poorly understood, with low compliance (Laveti and MacKay, 2009). Jit (2007) notes that the green turtle nesting beaches of Heemskereq Reefs and Ringgold Isles are vulnerable to illegal harvest by fishing vessels.

In Kiribati (e.g., Phoenix Islands), an unknown number of turtles are caught as bycatch on longlines and eaten (Obura and Stone, 2003). Poaching has been reported for Caroline Atoll, but to what extent it currently occurs is unknown (Teeb’aki, 1992).

In Tonga, Bell et al. (1994) report that collection of eggs for subsistence occurs, and Prescott et al. (2004) and Havea and MacKay (2009), also note that it is still a practice on islands where turtles nest. Bell et al. (2009) report that in Tonga sea turtles are harvested and live turtles are often seen transported from outer islands to the main island, Tongatapu. In 2007, Havea and MacKay (2009) conducted a survey in the three islands of Ha’apai to determine how many turtles were captured. They found that fishermen captured 56 turtles on O’ua, 23 on Ha’a’fefa, and 119 on Tungua. It is not clear how representative these three villages are for Ha’apai (another 7 islands or villages in Ha’apai were identified as hunting turtles in 1972). It is likely that this number is the minimum number of turtles captured in the Ha’apai Group (Havea and MacKay, 2009). No other data was reported on turtle hunting for other islands. Turtles were primarily captured by diving (hand), spear, and net, and used for consumption at home, local sales or barter, traditional occasions, and in some cases for a commercial market in the main island of Tongatapu (Havea and MacKay, 2009). It is unclear if this harvest is sustainable, especially given the increased catch rates in Tungua for the commercial market (Havea and MacKay, 2009).

In Tuvalu, harvest of sea turtles for their meat has been cited as a major threat (Alefaio and Alefaio, 2006).

In Tokelau, Balazs (1983) reported human take of both sea turtle eggs from nests and males and females while copulating, while nesting, or by harpoon. Apparent reductions in sea turtle numbers brought into question the sustainability of harvest in Tokelau and elicited discussion regarding conservation measures for the sea turtle population (Balazs, 1983). However, it appears sea turtles are still consumed in Tokelau (Ono and Addison, 2009).

In the Cook Islands, turtles are sometimes killed during nesting at Palmerston and Rakahanga, while nesting and via fishing on Nassau, and while nesting at Manihiki and Tongareva, and probably at other atolls; the exact level of take overall is unclear (White, 2012). At Tongareva (2011-2014) four females were taken while nesting, two juveniles and one adult female by net, one sub-adult speared, and four sub-adults (3 female, 1 male) were taken inwater by hand (White 2012; M. White unpubl.data). Turtles are occasionally speared underwater at Rakahanga (White and Galbraith, 2013). Only one clutch of eggs has been harvested at Tongareva Atoll during the last four years (2010-2014, M. White, unpubl.data). Take of turtles and eggs has been identified at Mauke, however the number taken is unknown (Bradshaw and Bradshaw, 2012).

Comment [A7]: White M, Galbraith GF (2013) Rakahanga Atoll: Sea turtles at a remote site in Oceania. Testudo 7: 30-48

RESPONSE: Revised text and included citation.
No information on overutilization could be found for the Territory of the Wallis and Futuna Islands.

13.2.5.3. Factor C: Disease or Predation

The extent and level of threat due to disease is not known in the South Central Pacific DPS. Depredation may have been a factor that contributed to the historical decline of this DPS. The best available data suggest that current nest and hatchling predation on several Central South Pacific DPS nesting beaches and in water habitats is a potential threat to this DPS.

As discussed above in this report, FP is the most commonly identified disease in green turtles and is characterized by the presence of internal and/or external tumors (fibropapillomas) that may grow large enough to hamper swimming, vision, feeding, and potential escape from predators (Herbst, 1994). While FP is recorded elsewhere in the Pacific (Van Houtan et al., 2010) it does not appear to be a threat in American Samoa (Utzurrum, 2002; A. Tagarino, American Samoa DMWR, pers. comm., 2013). Reports from other areas of this DPS are also unknown.

Polynesian Rats (Rattus exulans) were an issue at Rose Atoll prior to a 1993 eradication (FWS, 2012), but no longer appear to be a problem. The main threat to wildlife on Rose Atoll is the introduction (or possible reintroduction) of exotic species (K. Van Houtan, NMFS, pers. comm., 2013). Crabs are reported to eat hatchlings at Rose Atoll (Balazs, 1993; Ponwith, 1990; Pendleton pers comm., FWS, 2013). Blacktip reef sharks (Carcharhinus melanopterus) have been observed eating hatchlings in waters off Rose Atoll (Graeffe, 1873; Sachet, 1954; Balazs, 1999). On Swains Island, feral pig activity has been documented and may be a threat to nests on the island (Tagarino and Utzurrum, 2010).

In Samoa, feral animal predation on turtle nests and eggs remains a threat (SPREP, 2012; J. Ward, Government of Samoa, pers. comm., 2013).

Predation of green turtles (e.g., by sharks) occurs in French Polynesia; however the extent of such predation is unknown.

Given their remoteness, very little research has been conducted in the Phoenix Islands of Kiribati. However, numerous species are known to predate on sea turtle hatchlings and eggs and are found on Canton Island (one area green turtles are known to nest). The ghost crab (Ocypode sp.) is a predator of both hatchlings and eggs and was found at Canton Island, as was the land hermit crab (Coenobita sp.) which has been observed attacking hatchlings at other beaches (Balazs, 1975). Balazs (1975) also recorded rodents, sea birds, and sharks that all may be predating on hatchlings. Introduced animals, including feral cats, rats, and feral pigs, are reported problems for wildlife (Teeb’aki, 1992) and may threaten green turtles on certain islands in Kiribati such as Kiritimati.

In Tokelau, identified predators that may constitute a terrestrial threat to turtles include hermit crabs (Coenobita spp.), ghost crabs (Ocypode spp.), Polynesian rats (Rattus exulans), frigate...
birds (*Fregata ariel*, *F. minor*), and reef herons (*Egretta sacra*) (Balazs, 1983). In the marine environment, sharks and other carnivorous fish (e.g., groupers) may prey on sea turtles.

Feral pigs, rats, crabs, possibly some sea birds, and large fish are potential predators of sea turtles in the Cook Islands (White, 2012). Frigate birds (*Fregata ariel*, *F. minor*) and feral pigs are present on Tongareva; sharks, groupers and trevallys are present on the reefs and in the lagoon (M. White, unpubl.data). Pigs are reported on Mauke, although their impact on sea turtles is unquantified (Bradshaw and Bradshaw, 2012).

Shark predation of green turtles has been recorded at Palmyra Atoll (Sterling *et al.*, 2013).

13.2.5.4. **Factor D: Inadequacy of Existing Regulatory Mechanisms**

Lack of regulatory mechanisms and/or adequate implementation and enforcement is a threat to the Central South Pacific DPS. With regard to habitat, regulatory mechanisms are apparently inadequate to curb a continued loss of nesting habitat and degradation of foraging habitat due to human activities and coastal development on populated islands of American Samoa, Samoa, Tonga, Tuvalu, Fiji, and the Cook Islands. Regulatory mechanisms are in place that should address direct and incidental take of Central South Pacific green turtles (see section 13.2.6. Summary of Existing Conservation Efforts, below); however, these regulatory mechanisms are insufficient or are not being implemented effectively. As stated in the discussion of Factor C, turtles continue to be harvested for food and shells, and are used in commercial, subsistence, and ceremonial capacities. Woodrom Rudrud (2010) suggests that traditional laws in Polynesia may have historically limited green turtle consumption to certain people (chiefs, priests) or special ceremonies. However, as the societies of this region have been affected by Western culture and modernization of traditions have been altered, they have lost their effectiveness in limiting negative effects of harvest on sea turtles. In the following section (Factor E), we describe the insufficiency of regulatory mechanisms in relation to several threats including incidental bycatch in fishing gear, boat strikes, port dredging, debris, national defense, toxic compounds, and climate change. In this section we note that, apart from the American Samoa longline fishery (which is closely regulated and monitored), we could not confirm at what level effective management and regulation of fishery bycatch is occurring throughout the Central South Pacific DPS.

Several regulatory mechanisms that apply broadly to green turtles regionally or globally apply to green turtles within the Central South Pacific (see section 13.2.6. Summary of Existing Conservation Efforts for a list of these, and Appendix 2 for a discussion of each). Hykle (2002) and Tiwari (2002) reviewed the value of some international instruments, which vary in their effectiveness. Often, international treaties 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, 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 a 2002 special issue of the Journal of International Wildlife Law and Policy: International Instruments and Marine Turtle Conservation (Hykle, 2002).
Although information is scarce for this DPS, we find that the inadequacy of existing regulatory mechanisms and their enforcement for overutilization for commercial and subsistence purposes is both a significant and immediate threat to this DPS.

13.2.5.5. Factor E: Other Natural or Manmade Factors

The Central South Pacific DPS of the green turtle is negatively affected by both natural and anthropogenic impacts such as incidental fishery bycatch, interactions with recreational and commercial vessels, marine pollution, climate change, and major storm events.

**Incidental Bycatch in Fishing Gear**

Incidental capture in artisanal and commercial fisheries is a significant threat to the survival of greens sea turtles throughout the Central South Pacific DPS. The primary gear types involved in these interactions include longlines and nets.

Incidental capture in line, trap, or net fisheries presents a threat to sea turtles in American Samoa (Tagarino, 2011). Subsistence gill nets have been known to occasionally catch green turtles. Additionally, longline fishing is considered a threat to Central South Pacific green turtles. The American Samoa longline fishery is closely regulated and monitored, and is expected to kill up to 14 green turtles annually. These turtles represent multiple haplotypes (Marshall Islands, Yap, American Samoa; Great Barrier Reef, Coral Sea, New Caledonia; Marshall Islands; Fiji; Guam, Palau, Marshall Islands, Yap, Northern Mariana Islands, Taiwan and Papua New Guinea; Yap, BBR, New Caledonia, Coral Sea, Timor Sea, and east Indian Ocean; Maison et al. 2010). It is unclear exactly how many Central South Pacific green turtles in the South Pacific Ocean are taken in other longline fisheries, however it is estimated that over 200 green turtles could be killed annually by longline fishing in just the part of the South Pacific around American Samoa bounded by 180° and 155° W longitude, and 3° S – 32° S latitude (Maison et al., 2010).

In Fiji, green turtles are killed in commercial fishing nets, however the exact extent and intensity of this threat is unknown (Rupeni et al., 2002). Jit (2007) suggests that sea turtle bycatch is occurring in tuna fisheries in Fiji, but no information is provided on possible extent of sea turtle take or the species that are possibly taken. However, McCoy (2008) reports that green turtle bycatch is occurring in longline tuna fisheries in Fiji. Unfortunately, fishing trips do not appear to properly represent spatial and temporal distribution of fishing effort throughout the year, and the level of observer coverage is low, so the exact level of interactions with green turtles is unclear.

In the Cook Islands, longline fishery regulations require fishers to adopt the use of circle hooks and to follow “releasing hooked turtles” guidelines (Cook Islands Marine Resources Longline Fishery Regulations, 2008), although it is unclear how effective these regulations are. McCoy (2008) suggests that sea turtle bycatch is occurring in tuna fisheries in the Cook Islands; however, no information is provided on possible extent of sea turtle take or the species that are possibly taken. White (2012) reports that Cook Islands territorial waters are fished by other countries; however, the extent of sea turtle bycatch has not been fully analyzed and is unclear.
Marine Debris and Pollution

The ingestion of and entanglement in marine debris is another anthropogenic threat to green turtles. Direct or indirect disposal of anthropogenic waste introduces potentially lethal materials into green turtle foraging habitats. Green turtles will ingest plastic, monofilament fishing line, and other marine debris (Bjorndal et al., 1994). Effects may be lethal or non-lethal, resulting in varying effects that may increase the probability of death (Balazs, 1985; Carr, 1987; McCauley and Bjorndal, 1999). As in other parts of the world, marine debris presents a threat to green turtles in American Samoa (Aeby et al., 2008; FWS, 2012; Tagarino et al., 2008). Marine debris is potentially hazardous to adults and hatchlings and is present in American Samoa at Rose Atoll (FWS, 2012). It is also a threat at nearby inhabited islands. For example, a green turtle from Tutuila was necropsied in 2007 and contained plastic and aluminum (Tagarino et al., 2008). The exact number of turtles affected by this threat annually is unknown.

Marine pollution can also affect green turtles and their habitats in both the neritic and oceanic zones. These impacts can include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as impacts on water quality (e.g., increases in water column sediments) resulting from structural degradation from excessive boat anchoring, dredging, and other sources (Francour et al., 1999; Lee Long et al., 2000; Waycott et al., 2005). Pago Pago Harbor is seriously polluted, and uncontrolled effluent contaminants have impaired water quality in some coastal waters (Aeby et al., 2008). Effects to coastal habitat (e.g., reefs) from sedimentation related development and runoff is a significant potential threat in American Samoa, and human population pressures place strains on shoreline resources (Aeby et al., 2008).

Ship groundings (e.g., at Rose Atoll in 1993) damaging reef habitat and spilling fuel and other contaminants, degradation of coastal waters due to silt-laden runoff from land and nutrient enrichment from human discharges and wastes, and heavy metal as well as other contaminant problems are threats to green turtles in American Samoa (FWS, 2012; NMFS and FWS, 1998).

In Fiji, pollution has been identified as a threat to sea turtles, however it is unclear how significant it is. Weaver (1996) identified potential threats to sea turtles from heavy metals and industrial waste, organic loadings in coastal areas, plastic bags, and leachate poisoning of sea grass foraging areas. Jit (2007) suggests that sea turtles in Fiji are threatened by pollution.

The exact extent of pollution in the Cook Islands is unclear, however White (2012) identified marine debris as a ubiquitous problem for sea turtles. He also noted possible issues with oil, tar, or toxic chemicals and terrestrial run-off into lagoons, especially at Rarotonga. Bradshaw and Bradshaw (2012) note pollution (e.g., accumulation of plastics on the beach) on Mauke, however it does not appear to be in quantities that would affect nesting turtles. Three beach cleans were undertaken at Tongareva in 2013; over 90 percent of the debris was plastic (M.White, unpubl.data. www.honucookislands.com ).
Climate Change

Climate change is another factor that has the potential to greatly affect green turtles. Potential impacts of climate change to green turtles include beach erosion from rising sea levels, repeated inundation of nests, skewed hatching sex ratios from rising incubation temperatures, and abrupt disruption of ocean currents used for natural dispersal during the complex life cycle (Fish et al., 2005, 2008; Hawkes et al., 2009; Poloczanska et al., 2009). Impacts from global climate change induced by human activities are likely to become more apparent in future years (IPCC, 2007a).

Tokelau, as an example, is very vulnerable to climate change and sea level rise owing partly to its small land mass surrounded by ocean, and its location in a region prone to natural disasters. The impact of climate change is expected to affect the physical and biological characteristics of the coastal areas, ecosystem structure and functioning. This will affect near-shore marine and coastal areas, many wetlands and mangroves and other trees by changes in sea level and storm surges. [http://www.sprep.org/Tokelau/tokelau](http://www.sprep.org/Tokelau/tokelau)

A recent study of 27 atoll islands in the central Pacific (including Kiribati and Tuvalu), for example, demonstrated that only 14% of islands decreased their area over a 19–60 year time span (Webb and Kench, 2010). This occurred in a region considered most vulnerable to sea-level rise (Nicholls and Cazenave, 2010) during a period in which sea-levels rose 2mm yr\(^{-1}\). While most islands maintained (43%) or increased in area (43%) all islands demonstrated significant morphological shifts, thus explaining the otherwise counterintuitive results. Though low-lying tropical islands are considered the most exposed to sea-level rise these historical data indicate that beach losses do not necessarily accompany sea-level increases.

Natural Disasters

Catastrophic natural environmental events, such as cyclones or hurricanes, may affect green turtles in the Central South Pacific Ocean. These types of events may disrupt green turtle nesting activity (Van Houtan and Bass, 2007), even if just on a temporary scale.

13.2.6. Summary of Existing Conservation Efforts

There are a number of islands and atolls in this DPS, spread across an expansive area. Conservation efforts, such as establishment of protected areas, exist that are beneficial to green turtles. It is unclear how well they and the national legislation relating to green turtles are working. It appears that the remoteness of some of the areas is providing the most conservation protection for certain threats.

13.2.6.1. National Conservation Legislation

American Samoa

Green turtles in American Samoa are currently fully protected under the ESA. The ESA has as its purpose 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. 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 conservation plan (often called HCP for 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. The ESA has been instrumental in curtailing the demise and assisting in the recovery of many species, and green turtles are included among them.

Green turtles are also protected by the Fishing and Hunting Regulations for American Samoa (24.0934) which prohibit the import, export, sale, possession, transport, or trade of sea turtles or their parts and take (as defined by the ESA) and carry additional penalties for violations at the local government level (Maison et al., 2010). Additionally, an American Samoa Executive Order in 2003 established the territorial waters of American Samoa as a sanctuary for sea turtles and marine mammals. It is not known how effective implementation of these protections are in American Samoa. For example, NMFS and FWS (1998) notes that the concept of conservation faces difficulties, many people are unaware that it is illegal to take turtles, and some believe that if they don’t take the turtles someone else will. To the degree they are effective, it is unclear to what extent territory protections would remain in place if the species were to be de-listed under the ESA.

In 2003, American Samoa declared its territorial seas a Whale and Turtle Sanctuary. At the national level, the NOAA National Marine Sanctuary of American Samoa is comprised of six protected areas, covering 13,581 square miles of nearshore coral reef and offshore open ocean waters across the Samoan Archipelago. The Sanctuary’s management plan includes action plans to help improve ecosystem-based management and help reduce existing and potential resource threats. Additionally, Rose Atoll Marine National Monument was established in 2009 and encompasses the Rose Atoll National Wildlife Refuge. These protected areas should provide some level of protection for green turtles and their habitat, however the effectiveness of these monuments for this species is unknown.

**Cook Islands**

No nationwide sea turtle legislation appears to exist in the Cook Islands; however Environment (Mitiaro, Atiu, and Takutea) Regulations 2008 exist for sea turtles (Cook Islands Environment Mitiaro Regulation 2008; Cook Islands Environment Atiu and Takutea Regulations 2008) for these specific areas within the Cook Islands. The regulations prohibit anyone from possessing, disturbing, killing, harming, removing or damaging any living wild turtle or eggs; disturbing nesting areas, nests, or removing any eggs from nests; exporting any turtle, eggs or parts thereof whether living or dead from Mitiaro, Atiu, or Takutea. However, it appears that there are exceptions for “traditional practice.” Longline fishery regulations require fishers to adopt the use
of circle hooks and to follow “releasing hooked turtles” guidelines (Cook Islands Marine Resources Longline Fishery Regulations 2008). It is unclear how effective these regulations are.

Suwarrow Atoll National Park includes nesting area for green turtles on Turtle Island (Pulea 1992).

The Northern Atolls (Pukapuka, Nassau, Rakahanga, Tongareva, and Manihiki to a lesser extent) are mainly subsistence cultures and use traditional methods and practices 'rahui' to manage their natural resources (White 2012). Rahui is decided and administered by each Atoll Council, and agreed to by public consent. When in place it is tapu to take those resources; at present sea turtles are not included, however, discussions with Councils are progressing well (M. White, pers. com; White and Galbraith 2013). A community-project has just begun at Tongareva to catalogue and manage its biodiversity (funders include NMFS, Rufford Trust, and SWOT; M. White pers. com). This project includes education, training local researchers, integrating scientific and traditional knowledge and practices, and translating scientific resources into the local language www.honucookislands.com

Fiji

Local traditional custom was formerly found in Fiji whereby “no take” zones were put in place for a period of time before significant festive occasions, and this tabu would be observed by all members of the clan of the chief. It is unclear if the tabu system is currently adhered to or applied to sea turtles. However, Fisheries (Protection of Turtles Amendment) Regulations 2010 exist to protect sea turtles. The Amendment was made under Section 9(g) of the Fisheries Act, and no person shall molest, take or kill turtles of any species; sell, offer, or expose for sale or export any turtle shell, flesh or derivatives; dig up, use, take or destroy turtle eggs of any turtle species; use turtle, turtle derivative, eggs or turtle shells for any purpose including education, research or tourism; or negatively impact turtle habitats. However, the Minister may exempt a person from these regulations (Nasome, 2010). These regulations expire on 31 December 2018 (Nasome, 2010). The effectiveness of these types of regulations in Fiji is unclear, given that compliance and enforcement of existing Fisheries legislation and regulations appears to be a challenge, especially in isolated communities, given limited resources of government agencies (Jit, 2007). Laveti and MacKay (2009) found that turtles continued to be harvested, despite prohibitions that existed during the 2004–2008 moratorium. Jit (2007) notes that no conservation measures have been taken for the remaining green turtle nesting beaches of Heemskerk Reefs and Ringgold Isles.

Efforts are being made to expand turtle monitoring and related protection and conservation efforts. While illegal harvest still remains a threat in Fiji, the Turtle Monitors network is working to improve the situation and further sea turtle conservation (http://www.wwfpacific.org.fj/?208573/Turtle-Monitors-Network-Expansion, May 11, 2013 entry).
**French Polynesia**

This country has established reserves for Scilly (Manuae) and Bellinghausen (Motu One), which include sea turtles and their protection (Petit, 2009; Maison *et al.*, 2010). The Sea Turtle Clinic (located in Moorea and created in 2004) and Turtle Center provide medical care to sick and injured sea turtles (http://www.temanaotemoana.org/conservation/the-sea-turtles-clinic/). Additionally, the Sea Turtles Observatory was created in 2011, with a mission to create educational tools and training sessions, and to implement new research and conservation initiatives for sea turtles (http://www.temanaotemoana.org/observation-networks/sea-turtles-observation-network/). It is unclear how effective any of these efforts are.

**Kiribati**

Wildlife Conservation Ordinance prohibits anyone to hunt, kill or capture any wild turtle on land and fully protects the green turtle on Birnie Island, Caroline Island (also known as Millennium Island), Christmas Island (also known as Kiritimati), Flint Island, Garner Island, Hull Island, Malden Island, McKean Island, Phoenix Island, Starbuck Island, Sydney Island, and Vostock Island (Maison *et al.*, 2010).

**Phoenix Islands Protected Area (PIPA)**

With a size of 408,250 km² (157,626 sq. miles) it is one of the largest marine protected areas in the world and the largest marine conservation effort of its kind by a Least Developed Country (LDC). Kiribati first declared the creation of PIPA in 2006 and on January 30, 2008, Kiribati adopted formal regulations for PIPA that more than doubled the original size to make it at that time the largest marine protected area on Earth. In 2010, PIPA was added to the list of UNESCO World Heritage sites. It is the largest and deepest World Heritage site on Earth. PIPA includes all eight atoll and low reef islands of the Kiribati section of the Phoenix Island group: Rawaki, Enderbury, Nikumaroro, McKean, Manra, Birnie, Canton and Orona. PIPA also includes two submerged reefs, Carondelet Reef and Winslow Reef, with Carondelet Reef being as little as 3 to 4 meters underwater at low tide. It is estimated that there could be more than 30 seamounts within PIPA, though to date only nine have been named. The greater part of PIPA by area is comprised of ocean floor with a water column averaging more than 4,000 meters (2.5 miles) deep with a maximum at 6,147 meters. Sea turtles and dolphins were observed at many of the islands, and evidence of turtle nesting was found on many of the beaches. Its remote location and protection are beneficial to the species. http://phoenixislands.org/.

**Pitcairn Islands**

Local Government Ordinance of 2001 states that no person may harass, hunt, kill or capture any sea turtle; however exceptions can be allowed for scientific purpose or traditional subsistence use (Maison *et al.*, 2010)
**Samoa**

Local fisheries regulations relevant to sea turtles (taken from Bell et al., 2010) are summarized here. Fishing Regulation 4 allows the government to declare a period or periods when fishing for green turtles is prohibited; however this has never been implemented. Fishing Regulation 7 prohibits fishing, possession, or sale of green turtles with a shell less than 700 mm (27.6 inches) curved carapace length. This has limited value as it allows harvest of large turtles that are important to population recovery. Fishing Regulation 7 also prohibits the disturbance of the nest of any turtle, or any person to take, use or sell or destroy the egg of any turtle. Marine Wildlife Protection Regulations require any person who accidentally captures, injures or kills a marine turtle whilst undertaking any fishing activity in Samoan waters to report the incident to the government (accidentally caught turtles must be released). Marine Wildlife Protection Regulation 8 makes it an offence to undertake any activity related to the commercial fishing of turtles, as well as take or catch turtles except when for subsistence and the taking is in accordance with requirements in relation to size, closed seasons, or any other matter. It also makes it illegal to take a female migrating to a beach between November 1st and the end of February, to take a female while laying eggs or on a nesting beach, to disturb eggs, to take or possess eggs, sell or purchase eggs, interfere with or disturb any nest, or export turtle shell or product without a permit. Laws also exist to control keeping turtles in captivity and exporting shell or turtle product without a permit. Regulation 9 requires tourism or turtle watching related activity conducted within the vicinity of turtles to be licensed and follow guidelines so as to not affect their movement or activities. Regulation 16 states no permit shall authorize the use of methodologies of scientific research into turtles that result in the death of sea turtles. While there are no specific traditions protecting turtles, villages can make rules concerning the harvest of any marine animals (it is unclear if this is occurring or is effective for green turtles). Unfortunately, due to limited resources and inadequate collaboration within the government, enforcement of legislation is ineffective (Bell et al., 2010).

The Safata and Aleipata Marine Protected Areas prohibit the harvesting of any turtles (Bell et al., 2010). While a national marine sanctuary was established in the Samoan EEZ for marine animals included turtles, a management plan/program for implementation has not been completed. Samoa is a member of groups such as SPREP, which has a marine turtle action plan (Bell et al., 2010).

**Tokelau**

Actual legal status of sea turtles is unclear. One source suggests that turtle fishing is prohibited in Tokelau, but also reports that it appears turtles are still eaten by some villagers. Passfield (1998) states that in Fakaofu there is a ban on taking turtles while nesting. Project GloBAL (2009f) reports that no regulations exist at the national level to protect sea turtles.

**Tonga**

Fisheries (Conservation and Management) Regulations 2006 specify that no person shall disturb, take, possess, sell or purchase turtle eggs; prohibit any person from interfering with or disturbing any turtle nest; prohibit using a spear or spear gun to take a turtle; prohibit harming or taking any
male turtle with a shell of less than 45 centimeters; prohibit harming or taking a male turtle during the closed season; prohibit sale of turtle meat out of the shell, unless certified it came from a turtle of legal size; prohibit harm or take of any female turtle; and establish a closed season for all male turtles from August to February, i.e., hunting is allowed March to July (Bell et al., 2009). Unfortunately, a number of sea turtle regulations are not being adhered to.

**Tuvalu**

Maison et al. (2010) reports that it is prohibited by Wildlife Conservation Ordinance 1975 to hunt, kill, or capture any wild turtle on land except if authorized by a valid license issued by the Minister. However, effectiveness of this ordinance is unclear.

**Wake, Baker, Howland, and Jarvis Islands, Kingman Reef, and Palmyra Atoll**

The Pacific Remote Islands Marine National Monument was established in January 2009 and is cooperatively managed by the U.S. Secretary of Commerce (NOAA) and the U.S. Secretary of the Interior (FWS), with the exception of Wake Island and Johnston Atoll, which are currently managed by the Department of Defense. National Wildlife Refuges also exist at each of the islands within the Monument. The areas extend 50 nautical miles from the mean low water lines and include green turtle habitat. The protected areas provide some protection to sea turtles and their habitat (e.g., through permitted access and no take protected areas) as well as their remoteness. [http://www.fpir.noaa.gov/MNM/mnm_prias.html](http://www.fpir.noaa.gov/MNM/mnm_prias.html)

**Wallis and Futuna**

No information could be found.

### 13.2.6.2. International Instruments

Several treaties and/or regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the Central South Pacific DPS. The mechanisms listed below apply to sea turtles found in the Central South Pacific DPS and are described in Appendix 5.

- Convention on the Conservation of Migratory Species of Wild Animals (CMS)
- Convention on Biological Diversity (CBD)
- Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention)
- Convention for the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean (WCPF Convention)
- Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)
- Convention for the Protection of the Natural Resources and Environment of the South Pacific Region (Noumea)
- Convention on Wetlands of International Importance (Ramsar)
- FAO Technical Consultation on Sea Turtle-Fishery Interactions
- Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC)
- International Convention for the Prevention of Pollution from Ships (MARPOL)
13.3. Assessment of Significant Portion of its Range (SPR)

The SRT reviewed the information on threats and extinction risk within each DPS to determine whether there were portions of the range of the DPS that might be considered “significant portions of the range” in accordance with the definition of endangered and threatened species and the draft policy that interprets that term (see Section 3.4).

In the Central South Pacific DPS, the nesting abundance is low but widespread throughout the DPS. Threats are likewise relatively uniform throughout the DPS, with the main threat being lack of enforcement and persistent low level subsistence harvest. The information on degree of threats and trends are limited; however, there is no reason to believe that there are portions of the range that are at substantially higher risk of extinction than others. The SRT concluded that the need to consider a significant portion of the range does not apply to this DPS.

13.4. Assessment of Extinction Risk

For the SRT's assessment of extinction risk for green turtles in the Central South Pacific DPS, there were two separate sets of ranking exercises: One focusing on the importance that each SRT member placed on each of the six critical assessment elements for this region (Table 13.3), and a second which reflects SRT members’ expert opinion about the probability that green turtles would fall into any one of the different extinction probability ranges (Table 13.4; see Section 3.3 for discussion of this process).
Table 13.3. Summary of ranks that reflect the importance placed by each SRT member on the critical assessment elements considered for the Central South Pacific DPS. See section 3.3. for details on the six elements and the voting process. For Elements 1–4, higher ranks indicate higher risk.

<table>
<thead>
<tr>
<th>Critical Assessment Elements</th>
<th>Element 1</th>
<th>Element 2</th>
<th>Element 3</th>
<th>Element 4</th>
<th>Element 5</th>
<th>Element 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abundance (1 to 5)</td>
<td>3.2</td>
<td>2.9</td>
<td>1.9</td>
<td>2.2</td>
<td>–1.3</td>
<td>0.5</td>
</tr>
<tr>
<td>Trends / Productivity (1 to 5)</td>
<td>0.3</td>
<td>0.3</td>
<td>0.2</td>
<td>0.3</td>
<td>0.2</td>
<td>0.2</td>
</tr>
<tr>
<td>Spatial Structure (1 to 5)</td>
<td>2–4</td>
<td>1–4</td>
<td>1–3</td>
<td>1–4</td>
<td>(–2)–0</td>
<td>0–2</td>
</tr>
<tr>
<td>Diversity / Resilience (1 to 5)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Five-Factor Analyses (–2 to 0)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Conservation Efforts (0 to 2)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

With respect to the important rankings for the six critical assessment elements (and considering the lack of long-term monitoring in this DPS and the subsequent lack of quantitative analyses thereof) nesting abundance featured most prominently in the risk threshold (mean score of 3.2 out of 5). The Population Trends / Productivity (mean score of 2.9) and Diversity / Resilience (mean score of 2.2) also featured significantly in the risk threshold voting, presumably due to the declining trend at largest nesting site and all rookeries of basically one type, as well as the general uncertainty from a lack of monitoring. SRT members considered future threats not yet experienced by the population as weighing heavier in their risk assessment voting (mean score of –1.3 of –2) than did any conservation efforts that may emerge in the future (mean score of 0.5 of 2). With respect to the diversity of opinions among the SRT members when considering the six critical assessment elements, the Trends / Productivity and Diversity / Resilience elements had the largest range, both ranging from 1 to 4. This spread of values may, again, reflect an uncertainty from the lack of monitoring in this DPS.

Table 13.4. Summary of Green Turtle SRT member expert opinion about the probability that the Central South Pacific DPS will reach a critical risk threshold within the next 100 years. Each SRT member assigned 100 points across all rank categories. The continuum in Row 1 has categories with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability Of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1-5%</th>
<th>6-10%</th>
<th>11-20%</th>
<th>21-50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN ASSIGNED POINTS</td>
<td>38.2</td>
<td>19.9</td>
<td>20.5</td>
<td>8.2</td>
<td>9.5</td>
<td>3.7</td>
</tr>
<tr>
<td>SEM</td>
<td>12.1</td>
<td>5.3</td>
<td>6.9</td>
<td>3.9</td>
<td>6.0</td>
<td>2.6</td>
</tr>
<tr>
<td>Min</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Max</td>
<td>95</td>
<td>50</td>
<td>65</td>
<td>40</td>
<td>50</td>
<td>25</td>
</tr>
</tbody>
</table>

Of the six critical risk threshold categories describing the probability that the Central South Pacific DPS will reach a critical risk threshold within 100 years (Table 13.4), the SRT member
votes resulted in the greatest probability designations in the ‘<1 %’ risk range, with a mean of 38.2 points. The next greatest probability designations were about even in the ‘1–5%’ (19.9) and ‘6–10%’ (20.5) categories. The categories with the fewest allocated points were the ‘>50%’ and ‘11–20%’ ranges, with means of 3.7 and 8.2 respectively. In their vote justifications, most members cited the low nesting abundance in this DPS as the primary factor influencing their vote.

Figure 13.2. Bar graph depicting Green Turtle SRT member expert opinion about the probability that the Central South Pacific DPS will reach a critical risk threshold within 100 years.

Additional factors that were cited included the persistent population threats of illegal harvesting (for subsistence and commercial purposes) and the looming effects of climate change. Voters suggested climate change (in particular sea level rise) may influence this DPS significantly as most nesting occurs at low-lying oceanic atolls. The overall lack of information was a concern mentioned by several members, and the reported declining trend at the largest nesting site was also a concern. Several members also referenced that such geographically dispersed and remote (from human population centers) nesting sites may actually favor future population persistence, insulating the population from some human-related threats. As a result, the vote justifications provided for this DPS were somewhat inconsistent across SRT members, depending on which factors they weighed as most significant.

13.5. Synthesis and Integration

The Central South Pacific DPS is characterized by geographically widespread nesting at low levels of abundance, mostly in remote low-lying oceanic atolls. Nesting is reported in 57
different locations. Chronic and persistent illegal harvest is a concern, as is sea level rise. Climate change is considered to perhaps affect this DPS more than any other. However, there are no globally significant human population centers (in terms of abundance) in this DPS, and most rookeries are remote and inaccessible from the major human population centers that lie within this DPS. There are also no long-term monitoring programs that have been active in this DPS, for even a 5-year period.

Despite the low overall abundance of nesting females and various population threats, SRT members attributed the largest probability (38.2) to the lowest single category of extinction risk (<1 percent). This is likely due to the broad expanse of nesting in 57 different sites and the lack of documented acute population threats from globally significant urban areas and human population centers. However, the characteristics of this DPS did lead voters to conclude a 62 percent probability of having a greater than 1 percent extinction risk, meaning that chronic harvesting and climate change are real and persistent threats over the next 100 years.
14. CENTRAL NORTH PACIFIC DPS (DPS #10)

14.1. DPS Range and Nesting Distribution

The Central North Pacific DPS covers the Hawaiian Archipelago and Johnston Atoll. It is bounded by a four-sided polygon with open ocean extents reaching to 41°N, 169°W in the northwest corner, 41°N, 217°W in the northeast, 9°N, 235°W in southeast, and 9°N, 185°W in the southwest. The Hawaiian Archipelago is the most geographically isolated island group on the planet and, therefore, it is perhaps unsurprising that green turtles in this DPS are geographically discrete in their range and movements, as evidenced by mark-recapture studies using flipper tags, PIT tags and satellite-linked transmitter tracking (Figure 14.1).

Figure 14.1. Geographic area of the Central North Pacific DPS. Size of circles indicates nesting abundance category. DPS encompasses the entire Archipelago of Hawai‘i and Johnston Atoll.

From 1965 to 2013, 17,536 green turtles have been tagged involving all post-pelagic size classes from juveniles to adults. With only three exceptions, the 7,360 recaptures of these tagged turtles have been made within the Hawaiian Archipelago. The three outliers involved a recovery in Japan, one in the Marshall Islands and one in the Philippines. French Frigate Shoals (FFS), located in the Northwest Hawaiian Islands (NWHI), represents the most prominent focal point of green turtle nesting and hatching production in the Hawaiian Archipelago (Figure 14.2). Information from tagging at FFS, other areas in the NWHI, areas in the Main Hawaiian Islands (MHI), and Johnston Atoll show that the vast majority of reproductive females and males periodically migrate to FFS for seasonal breeding from these distant locations. At the end of the season, they return to their respective foraging areas. Conventional tagging using PIT and metal

Comment [A1]: Comment: Maybe also refer to their genetic isolation (Dutton et al 2008)
Comment [A2]: Response: revised
Comment [A3]: Response: Don’t think we need “most”. It is my understanding that it is the prominent focal point.
flipper tags have documented 164 turtles making reproductive movements from or to FFS and foraging pastures in the MHI, and 58 turtles from or to FFS and the foraging pastures in the NWHI.

Figure 14.2. Nesting distribution of green turtles in the Central North Pacific DPS. Size of circles indicates nesting abundance category.

As stated, the principal nesting site for green turtles in the Central North Pacific DPS since 1960 is FFS (Balazs, 1980; Lipman and Balazs, 1983); where 96 percent of the population currently nests. However, nesting was historically abundant at various sites across the archipelago as recently as 1920 (Kittinger et al., 2013). Within FFS, East Island accounts for ~50 percent of nests, while other islets of FFS—Tern, Trig, Gin, and Little Gin—account for the remainder. Whale-Skate, joined by sand deposition between the former islets of Whale and Skate in the 1950s, eroded and became submerged in 1997 (Baker et al., 2006). Nesting by green turtles occurs in low numbers throughout the NWHI at Laysan, Lisianski, Pearl and Hermes Reef, and very uncommonly at Midway and Kure. Green turtle nesting has recently emerged in the MHI as well (Dutton et al., 2008; Frey et al., 2013) albeit at low abundance (1-2 nesters annually) at numerous, widespread locations.

Green turtles in the Central North Pacific DPS bask on beaches throughout the NWHI and in the MHI. Basking in reptiles is considered to affect thermoregulation (Bartholomew, 1965; Whittow and Balazs, 1982), raise core body temperatures (Swimmer, 2006), and has been anecdotally observed to vary seasonally in Hawai’i (K. Van Houtan, NMFS, pers. comm., 2013).
14.2. Critical Assessment Elements

In the evaluation of extinction risk for green turtles in the Central North Pacific, the SRT considered six critical assessment elements, each of which are discussed below: (1) Nesting Abundance, (2) Population Trends, (3) Spatial Structure, (4) Diversity / Resilience, (5) Five-Factor Threat Analyses, and (2) Existing Conservation Efforts. See Section 3.3 for additional information on the selection of these six critical assessment elements.

14.2.1. Nesting Abundance

Table 14.1 shows the level of nesting occurring in this DPS with an estimated 3,710 breeding females, the green turtle nesting concentration at FFS (Hawai'i, USA).

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Hawai'i (USA)</td>
<td>Lana'i</td>
<td>2010-2012</td>
<td>1</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>Kaho'olawe</td>
<td>2010-2012</td>
<td>1</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>Hawai'i</td>
<td>2010-2012</td>
<td>1</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>Midway Atoll</td>
<td>2011</td>
<td>3</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>Maui</td>
<td>2010-2012</td>
<td>4</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>O'ahu</td>
<td>2010-2012</td>
<td>11</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>Lisianski Island</td>
<td>2011</td>
<td>15</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>Kaua'i</td>
<td>2010-2012</td>
<td>16</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>Laysan Island</td>
<td>2011</td>
<td>24</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>Molokai</td>
<td>2011</td>
<td>24</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>Pearl Hermes Reef</td>
<td>2011</td>
<td>36</td>
</tr>
<tr>
<td>Hawai'i (USA)</td>
<td>French Frigate Shoals</td>
<td>2009-2012</td>
<td>3,710</td>
</tr>
</tbody>
</table>

Green turtle nesting in the Central North Pacific DPS is remarkably concentrated geographically (Table 14.2). More than 96 percent of the females in this DPS nest at FFS, with approximately 50 percent of those nesting on East Island. Each of the remaining 11 nesting sites in this DPS has fewer than 40 females in their nesting population. In sum, we estimate the DPS has a total of 3,846 breeding females. The DPS is geographically and chronologically well-sampled; there are no sites where nesting is unquantified, and there is very little chance there are undocumented nesting locations. The Central North Pacific has 12 nesting sites, the smallest such number globally.
Table 14.2. The distribution of green turtle nester abundance in the Central North Pacific.

<table>
<thead>
<tr>
<th>NESTER ABUNDANCE</th>
<th># NESTING SITES</th>
</tr>
</thead>
<tbody>
<tr>
<td>n/a</td>
<td>0</td>
</tr>
<tr>
<td>1-10</td>
<td>5</td>
</tr>
<tr>
<td>11-50</td>
<td>6</td>
</tr>
<tr>
<td>51-100</td>
<td>0</td>
</tr>
<tr>
<td>101-500</td>
<td>0</td>
</tr>
<tr>
<td>501-1000</td>
<td>0</td>
</tr>
<tr>
<td>1001-5000</td>
<td>1</td>
</tr>
<tr>
<td>5001-10000</td>
<td>0</td>
</tr>
<tr>
<td>10001-100000</td>
<td>0</td>
</tr>
<tr>
<td>&gt;100,000</td>
<td>0</td>
</tr>
<tr>
<td>TOTAL SITES</td>
<td>12</td>
</tr>
<tr>
<td>TOTAL ABUNDANCE</td>
<td>3,846</td>
</tr>
<tr>
<td>PERCENT at LARGEST NESTING SITE</td>
<td>96 percent</td>
</tr>
</tbody>
</table>

14.2.2. Population Trends

Since nesting surveys were initiated in 1973, there has been a marked increase in annual green turtle nesting at East Island, FFS, where approximately 50 percent of the nesting on FFS occurs (Balazs and Chaloupka, 2004b, 2006). During the first 5 years of monitoring (1973-1977), the mean annual nesting abundance was 83 females, and during the most recent 5 years of monitoring (2009-2012), the mean annual nesting abundance was 464 females (Balazs and Chaloupka, 2006; G. Balazs, NMFS, unpublished data). This increase over the last 40 years corresponds to an annual increase of 4.8 percent.

Information on in-water abundance trends is consistent with the increase in nesting (Balazs et al., 1996, 2005; Balazs, 2000). This linkage is to be expected since, based on genetics, satellite telemetry, and direct observation, green turtles from the nesting beaches in the FFS nesting site remain resident to foraging pastures throughout the archipelago (with the possible exception of the oceanic juvenile phase, for which there are no available data and which genetic sampling has yet to reveal) and are the exclusive nesting population present in these areas (Balazs, 1976; Craig and Balazs, 1995; Keuper-Bennett and Bennet, 2002; P. Dutton, NMFS, pers. comm., 2013). A significant increase in catch per unit effort of green turtles was seen from 1982 to 1999 during bull-pen fishing conducted at Palaau, Moloka'i (Balazs, 2000). The number of immature green turtles residing in foraging areas of the eight MHI has increased (Balazs et al., 1996). In addition, although the causes are not totally clear, there has been a dramatic increase in the number of basking turtles in the Hawaiian Islands over the last two decades, both in the southern foraging areas of the main islands (Balazs et al., 1996) as well as at northern foraging areas at Midway Atoll (Balazs et al., 2005). Although it is not possible to unequivocally tie this increase
in basking to an increase in total population abundance, it is possible that the increase in basking turtles reflects the increase in abundance of green turtles in the Central North Pacific DPS.

Population Viability Analysis was one component of the Population Trend element and was conducted for nesting sites that had a minimum of 15 years of recent nesting data with an annual nesting level of more than 10 females (for more on data quantity and quality standards used, see Section 3.2). To assist in interpreting these PVAs, we indicate the probability of green turtle nesting populations declining to two different biological reference points, one using a trend-based and the other an abundance-based threshold. The trend-based reference point for evaluating population forecasts is half of the last observed abundance value, i.e., a 50 percent decline. The abundance-based reference point was a total adult female abundance of 300 females. Risk is calculated as the percentage of model runs that fall below these reference points after 100 years. Nesters at East Island were counted directly, and observed data is a running sum of the calendar year plus the three previous years, given that females at this nesting site have a four year breeding interval. According to this model, the average growth rate \( r \) for this annual nesting abundance time series is 0.048 and the standard deviation is 0.142.

**Figure 14.3.** Stochastic Exponential Growth (SEG) Model Output for East Island, French Frigate Shoals, USA. Black line is observed data, green line is the average of 10,000 simulations, light green lines are the 2.5th and 97.5th percentiles (serving as the 95 percent credible interval), gray dotted line is trend reference, and red dotted line is absolute abundance reference.

For this population, the outputs of the PVA model based on 38 years (1975-2012) of nesting beach monitoring data indicate that there is 0 percent probability that this population will fall below the trend reference point (50 percent decline) at the end of 100 years, and a 0 percent probability that this population falls below the absolute abundance reference (100 females/yr) at the end of 100 years. This indicates that, based on performance of this population, and in the

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Comment [A10]: Comment: This is the sort of comment needed above - with some explanatory notes in the intro chapters.

Comment [A11]: Response: Noted
absence of external drivers other than those which currently exist, it is expected to continue to increase and remain above both biological reference points discussed above. However, it should be noted that this PVA modeling has important limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. For a full discussion of these PVAs and these reference points, see Section 3.2.

While the nesting population trajectory is positive and encouraging, more than 96 percent of nesting occurs at one site in the NWHI and it is highly vulnerable to threats. Survival of this DPS is currently highly dependent on successful nesting at FFS (Niethammer et al., 1992). The concentrated nature and relatively small size of the nesting population make it vulnerable to random variation and stochasticities in the biological and physical environment, including natural catastrophes, as well as changes in climate and resulting effects such as sea level rise. This increases its risk of extinction, even though it may have positive population growth (e.g., Hunter and Gibbs, 1997; Meffe et al., 1994; Primack, 1998; Balazs and Kubis, 2007). Both non-stochastic as well as stochastic events are significant current and future threats to this small, isolated, concentrated population.

14.2.3. Spatial Structure

When examining spatial structure for the Central North Pacific DPS, the SRT examined three lines of evidence, including genetic data, flipper and satellite tagging, and demographic data.

Green turtles in the Central North Pacific DPS have been studied extensively for decades and as a result there is a multitude of information on this population. Flipper tag returns and satellite tracking studies demonstrate that post-nesting females in the NWHI return to their foraging grounds in the MHI, and that foraging remains exclusively within geographic boundaries of this DPS. Demographic studies of green turtles do not reveal any structuring of traits within the DPS, although variable ecosystem productivity has produced differences in body conditions of nearshore foraging turtles (Balazs and Chaloupka, 2004a; Wabnitz et al., 2010). Nesters at the primary nesting area of FFS average 92.2 cm SCL, have an inter-nesting interval of 13.2 days, clutch size of 92.4 eggs, and a clutch frequency of 4 nests (Balazs, 1980; Niethammer et al., 1997; Tiwari et al., 2010). Peak nesting in this DPS occurs from May through August (Balazs, 1980), and nesters return to breed at an interval of 4 years (G. Balazs, NMFS, pers. comm., 2013). Nest-level inventories are incomplete for this DPS, however, Balazs (1980) estimated hatchling emergence at 71 percent. Though previous estimates of age at first reproduction were as high as 35-50 years (Balazs, 1980; Zug et al., 2002), a recent study examining capture histories, skeletochronology, and the observed nesting time series estimated green turtles in this DPS begin breeding on average at 23 years (K. Van Houtan, NMFS, pers. comm., 2013). As a result of the unusual nesting concentration at one site, typically studied population variables such as mean nesting size, nesting season, inter-nesting interval, clutch size, hatching success, nesting season, and clutch frequency have not been compared among multiple nesting sites within this DPS.
Genetic sampling in the Central North Pacific DPS has been extensive and representative given that there are few nesting populations in this region. Results of mtDNA analysis indicate a low level of spatial structure with regard to minor nesting around the MHI and the NWHI although the same haplotypes occur throughout the DPS. Within the NWHI, studies show no significant differentiation (based on mtDNA haplotype frequency) between FFS and Laysan Island (P. Dutton, NMFS, pers. comm., 2013). Frey et al.’s (2013) analysis of low level of scattered nesting on the MHI (Moloka‘i, Maui, O‘ahu, Lana‘i & Kaua‘i; mtDNA and nDNA) showed that nesting in the MHI might be attributed to a relatively small number of females that appear to be related to each other, and demographically isolated from FFS. Frey et al. (2013) suggest that the nesting population at the MHI may be the result of a few recent founders that originated from the FFS breeding population. This regional range expansion may buffer against the loss of current nesting sites at FFS due to sea level rise.

The current nesting distribution represents a curtailment of nesting activities within the Central North Pacific DPS’s historic range (see next section 14.2.4 below for more discussion).

14.2.4. Diversity and Resilience

The aspects considered under this critical element include the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation, and the genetic diversity within the DPS. Aspects such as these are important considerations for assessing the potential impact of catastrophic events such as storms, sea level rise, and disease.

As nesting in the Central North Pacific DPS is unusually concentrated at one site (Table 14.2) there is little diversity in nesting areas. Remnant nesting aggregations may have existed in the MHIs as recently as the 1930s, but were no longer present in the 1970s (Balazs, 1976). For example, an important green turtle nesting colony once nested on Lana‘i, and several select beaches on Moloka‘i, O‘ahu, and Kaua‘i were also used by green turtles (Balazs, 1975). Balazs (1980) reported that the distribution of green turtles in the Hawaiian Archipelago has been reduced within historical times. A more recent study (Kittinger et al., 2013) supports this finding and further suggests that there has been a significant constriction in the spatial distribution of important reproduction sites, presenting a challenge to the population’s future and making this DPS highly vulnerable. As much as 80 percent of historically major nesting populations could be extirpated or have heavily reduced nesting abundances, and what was once geographically distributed nesting is now concentrated at a single site (Kittinger et al., 2013).

The one nesting site, FFS, is a low-lying coral atoll that is susceptible to erosion, geomorphological changes and sea level rise, and has already lost significant nesting area (Baker et al., 2006). However, aside from sea level rise, FFS is relatively isolated from anthropogenic threats. The entire NWHI, which represents more than 98 percent of nesting in this DPS, lies within the Papahānaumokuākea Marine National Monument, a remote Monument that has controlled access for activities that occur within it.

Mitochondrial DNA studies indicate that there is a low level of stock substructuring among nesting sites in the Central North Pacific DPS (see Section 6.2.3) and a relatively low level of
14.2.5. Analysis of Factors Listed Under ESA Section 4(a)(1)

Section 4(a)(1) of the ESA lists five factors (threats) that must be considered when determining the status of a species. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

The following information on threats pertains to green turtles of the Central North Pacific DPS. Because foraging green turtles or migratory routes of green turtles from this DPS remain within the geographic boundary of this DPS, no other DPS narratives need to be consulted. Additionally, because the vast majority of this DPS lies within State of Hawai‘i and the entire DPS lies within the U.S., much is known of the threats, conservation efforts, and enforcement in the Central North Pacific.

14.2.5.1. Factor A: Destruction or Modification of Habitat or Range

Coastal development and construction, vehicular and pedestrian traffic, beach pollution, tourism, and other human related activities is an increasing threat to the basking and nesting population in the MHI (currently very limited) and may negatively affect hatchling and nesting turtles on these beaches. Climate change is a threat on the terrestrial and neritic/oceanic zone in both the MHI and NWHI and is expected to affect all life stages of green turtles.

**Terrestrial Zone**

Historically, impacts to nesting habitat in the MHI included land development, vehicle operation on beaches, alteration of native vegetation, and erosion (Balazs, 1975). Coastal development and construction, vehicular and pedestrian traffic, beach pollution, tourism, and other human related activities are current threats to nesting and basking habitat in the MHI. These threats will affect more green turtles in this DPS if nesting increases in the MHI. Human populations are growing rapidly in many areas of the insular Pacific, including Hawai‘i, and this expansion is exerting increased pressure on limited island resources. The human population of the MHI is nearly 1.4 million and growing. Millions of tourists visit Hawai‘i each year (e.g., almost 7 million in 2010; http://www.hawaiitourismauthority.org/research/reports/historical-visitor-statistics/) and many of them spend time in the terrestrial and marine habitats of sea turtles.

Most nesting currently occurs in the NWHI within the Papahānaumokuākea National Monument, where there is little human interaction with nesting turtles. However, this was not always the case. In the first half of the 1900s, military exercises were regularly conducted at FFS and Pearl and Hermes Reef in the NWHI, and a military air station was constructed on Tern Island at FFS.
that destroyed a significant amount of green turtle nesting habitat (Balazs, 1975). Balazs and Chaloupka (2004b) note that extensive nesting habitat damage occurred at FFS before the 1950s. Today, nesting on Tern Island is precluded for the most part by a sea wall that runs the length of one side of the island. Likewise, seawall construction at Johnston Atoll has preempted use of beaches by sea turtles (NMFS and FWS, 1998). East Island, where approximately half of FFS nesting now occurs, is accessible to turtles.

Climate change, including changing storm dynamics and intensity, and loss of nesting habitat, are emerging concerns for habitat in both the MHI and the NWHI (Baker et al., 2006; Keller et al., 2009). Weather events, such as storms and seasonal changes in current patterns, can also reduce or eliminate sandy beaches, degrade turtle nesting habitat, and cause barriers to adult and hatchling turtle movements on affected beaches.

Climatic changes in the NWHI pose threats through reduction in area of nesting beaches critical to this DPS (Baker et al., 2006). Baker et al. (2006) examined the potential effects of sea level rise in the NWHI and found that the primary nesting area for the Central North Pacific population is threatened by sea level rise through possible loss of nesting habitat. For example, Whale-Skate Island at French Frigate Shoals was formerly a primary green turtle nesting site for this DPS but the island has subsided and is no longer available for nesting (Kittinger et al., 2013). Trig, Gin and Little Gin could lose large portions of their area, concentrating nesting even further at East Island (Baker et al., 2006). Additionally, habitat degradation resulting from the release of contaminants contained in landfills and other areas of the NWHI could occur as the islands erode or are flooded from sea level rise (Keller et al., 2009). Effects of climate change are discussed further below in Section 14.2.5.5.4.

Neritic/Oceanic Zone

Threats to green turtle habitat in neritic and/or oceanic zones of the Central North Pacific DPS include contamination and degradation of foraging areas due to nearshore development, land-based sources of marine pollution and increased human activity, contamination due to past military practices and potential vessel groundings, fishing practices (see section 14.2.5.5), and climate change.

Impacts to the quality of coastal habitats in the MHI are a threat to this DPS and are expected to continue and possibly increase with an increasing human population and annual influx of millions of tourists. Loss of foraging habitat or reduction in habitat quality in the MHI due to nearshore development is a threat to this DPS. Marina construction, beach development, siltation of forage areas, contamination of forage areas from anthropogenic activities, resort development or activities, increased vessel traffic, and other activities are all considered threats to this population and its habitat (Bowen et al., 1992; NMFS and USFWS, 1998; Wedding and Friedlander, 2008; Wedding et al., 2008; Van Houtan et al., 2010; Friedlander and Brown, 2006). Sea grass and coral reef habitat of Moloka'i has been degraded from upland soil erosion and siltation; coral reefs of Hawai'i, Kaua'i, Lanai, Maui, and O'ahu have been degraded by sedimentation, sewage, or coastal construction (NMFS and FWS, 1998). In general, MHI coral reefs have suffered from land-based sources of pollution, overfishing, recreational overuse, and alien and invasive species (Friedlander et al., 2006), and are threatened by climate change and
increased temperatures resulting in coral bleaching events, coral disease, coastal development and runoff, and waste water (point source and non-point source pollution) (Friedlander et al., 2008). Climate change influences on water temperatures, ocean acidification, sea level and related changes in coral reef habitat, wave climate and coastal shorelines are expected to continue (Friedlander et al., 2008).

Vessel groundings (mechanical damage to habitat and reef-associated organisms) and related release of contaminants (e.g., fuel, hazardous substances, etc.) are a threat to Central North Pacific green turtle habitat (Keller et al., 2009). It is difficult to predict the exact number or severity of vessel groundings expected in any future year, however key nesting and foraging habitat for green sea turtles occurs in the areas of the MHI and the NWHI where commercial and recreational boating occurs. Thirteen reported vessel groundings have occurred in the NWHI in the last 60 years (Keller et al., 2009); vessel activity occurring in or around green sea turtle foraging habitat in the MHI islands is much greater than in the NWHI.

During the last century, Johnston Atoll was affected by human activities, including military activities such as nuclear testing and chemical weapons incineration. The lingering effects of these activities include water contamination from nutrients, dioxins, plutonium, and a subsurface plume of PCB-contaminated petroleum product (FWS, 2013).

Climate change may result in future trophic changes, thus affecting green turtle foraging and/or distribution. Elevated sea surface temperatures already appear to be affecting Central North Pacific ocean habitat (e.g., coral bleaching) in the NWHI. Consequences of climate change can vary between different life stages and can be potentially positive or negative, and it is challenging to predict the exact future magnitude of climate change and associated impacts or the adaptive capacity of this species. However, certain negative impacts (e.g., to reef habitat) are already occurring and are likely to intensify; the environment is expected to become more uncertain and potentially more risky for this species in the face of climate change. Climate change has been recognized as a potential threat to sea turtles, including neritic/oceanic habitat of the NWHI and the MHI in the Central North Pacific, and is discussed further below in Section 14.2.5.5.

14.2.5.2. Factor B: Overutilization

The harvesting of eggs and turtles was likely a factor that contributed to the historical declines of the population. Current illegal harvest of green turtles for human consumption continues in a limited way; however federal and state cooperative efforts and existing legislation in place appear to be minimizing the threat from illegal harvest.
Recent studies (Kittinger et al., 2011, 2013; Van Houtan et al., 2012) describe three distinct phases of sea turtle exploitation in the Central North Pacific DPS. The first phase was during indigenous Polynesian societies (1250-1778), the second between European contact and World War II (1779-1945), and the final phase between World War II and when federal and state protections began (1946-1974). These phases comprise different threats at varying magnitudes, affecting different segments of the population across its geographic range. Archeological excavations, for example, indicate hunting pressure from indigenous Polynesians was widespread and probably extirpated important nesting areas in the MHI (Kittinger et al., 2013).

In the 1800s, ships from Europe, North America, and Asia visiting the uninhabited NWHI frequently made large turtle harvests for subsistence and commercial trade (Elschner, 1915; Amerson, 1971; Kittinger et al., 2011; Van Houtan et al., 2012). By 1900, green turtles were ubiquitous in Honolulu markets and restaurants, and by 1950 nesting was essentially extirpated everywhere except on a single remote atoll.

From 1948 to 1974, commercial fishermen needed licenses and were required to file catch reports when harvesting green turtles (Van Houtan and Kittinger, 2014). The cumulative reported harvest during this period from both subsistence and commercial takes was 5,000-6,000 turtles—roughly 180-230 turtles a year. However, such annual totals during this period reportedly occurred previous to this period in Hawai'i during a single day (Clapp and Wirtz, 1975; Amerson et al., 1974; Kittinger et al., 2011; Van Houtan et al., 2012). While the managed commercial fishery for green turtles was small in scale in Hawai'i— with a limited effort, productivity, and revenue—there were dramatic declines in catch per unit effort and a spatial progression in the fishery that strongly suggest rapid local population depletion (Van Houtan and Kittinger, in press). Harvests initially targeted coastal areas near commercial markets but sequentially shifted to exploit more remote areas, expanded effort, and increasingly relied on more extractive gears. Additional analyses of economic data, restaurant menus, and expert interviews indicate the Hawai'i commercial green turtle fishery was driven by limited, local demand (Van Houtan and Kittinger, in press). The seemingly incommensurate scale of the fishery and its impacts indicate the Hawaiian green turtle population was already significantly depleted by World War II, when commercial fishery began, pointing to the significant prior exploitation (Kittinger et al., 2013).

Harvest of green turtles has been illegal since green turtles were listed under the ESA in 1978. It is possible that human take today is underreported, as anecdotal information suggests that some degree of illegal take occurs throughout the MHI, especially on the islands of Hawai'i, Maui, Kaua'i, Moloka'i and Ni'ihau. However, the exact extent of such take is unknown.

14.2.5.3. **Factor C: Disease or Predation**
The FP disease affects green turtles found in the Central North Pacific Ocean (Francke et al., 2013). This disease results in internal and/or external tumors (fibropapillomas) that may grow large enough to hamper swimming, vision, feeding, and potential escape from predators. In 2012 alone, 36 green turtle strandings involved FP (Francke et al., 2013). Due to limitations of stranding data, the exact numbers are unknown as reported strandings are an unknown fraction of all green turtle mortalities. FP appears to have peaked in some areas of Hawai‘i, remained the same in some regions, and increased in others (Van Houtan et al., 2010). Environmental factors may be significant in promoting FP, and eutrophication (increase in nutrients) of coastal marine ecosystems may promote this disease (Van Houtan et al., 2010). FP remains an important concern. This is particularly true given the continued, and possibly future increasing, human impacts to, and eutrophication of, coastal marine ecosystems that may promote this disease. Spirochid (blood fluke) infections are reported for Central North Pacific green turtles (Greenblatt et al., 2005; Work et al., 2005). However, the extent to which this is a threat to the population is unknown.

Predation of hatchlings is well known for green turtles. Sea turtle ecology and biology encompasses natural predation and green sea turtles have evolved with it. Predation by some native species is considered a normal part of their life history; however, predation may be problematic when it involves additional non-native species and to the extent that it exerts additional pressure on the population when considered in the context of additional anthropogenic sources. Ghost crabs (Ocypode spp.) prey on hatchlings at FFS (Niethammer et al., 1997). Ghost crabs (Ocypode spp.) prey on hatchlings at FFS (Niethammer et al., 1997). The exact number of hatchlings lost to this depredation is not known, but is estimated at approximately 5 percent (Balazs, 1980). Hatchlings may also be eaten by fish when they enter the marine environment. Large grouper (Epinephelus tauvina) are documented predators of post-hatchling green turtles in Hawai‘i; however, the extent of grouper predation is unknown (Balazs, 1979). Sea birds may also prey on sea turtles in the marine environment (Balazs and Kubis, 2007). Less natural to Hawai‘i are mongoose, rats, dogs, feral pigs, and cats—all introduced species—that exist on the MHI and are known to prey on eggs and hatchlings, although the exact impact on the current low level of nesting is unclear (nesting in the MHI is extremely low compared to historical levels). If nesting in the MHI increases, the importance of the threat from these potential predators would increase.

Stranding records of the Hawaiian Islands (e.g., Francke, 2013) show shark depredation of Central North Pacific green turtles. The exact numbers of animals taken by sharks is unknown, as reported strandings only represent a fraction of all green turtle mortalities. Balazs (1979) suggests that depredation by sharks on both adults and immature turtles could be substantial for green turtles throughout Hawai‘i. Stranding records of the Hawaiian Islands (e.g., Francke et al., 2013) show shark depredation of Central North Pacific green turtles. The exact numbers of animals taken by sharks is unknown, as reported strandings only represent a fraction of all green turtle mortalities. Balazs (1985) also describes the presence of sharks at Johnston Atoll and depredation could be a threat for green turtles at this Atoll, although detailed information on the extent of this possible threat at this location is unavailable.
14.2.5.4. Factor D: Inadequacy of Existing Regulatory Mechanisms

Regulatory mechanisms are in place that should be designed to address direct and incidental take of green turtles in the Central North Pacific DPS. These mechanisms include international laws and agreements such as the Convention on the Conservation of Migratory Species of Wild Animals (CMS) and the Convention for the Conservation of Migratory Species of Land Birds (CMS). Additionally, the State of Hawaii has enacted state-level regulations to protect green sea turtles, such as the Hawaii Coastal Erosion Management Plan.

Regulatory mechanisms that protect green turtles are in place and include state, federal, and international laws. A commercial ban and enforcement of turtle harvest was put into place by the State of Hawaii in 1974, 4 years before the green turtle was listed under the ESA in 1978. The additional conservation actions, as well as federal resources, that occurred when the green sea turtle was listed under the ESA has provided for comprehensive protection and recovery activities that have been sufficiently effective for the population to grow. Since listing under the ESA, protection has been sufficiently effective for the population to grow significantly; however, it is unclear the extent to which protections of the state law would continue if green turtles were delisted under the ESA. In addition, some threats to the species, such as climate change, are either not able to be regulated or not regulated sufficiently to control or even slow the threat. Others, such as bycatch in international fisheries, are not adequately regulated, although bycatch in domestic Federal fisheries has been addressed to a greater extent. See section 14.2.5.5 (Factor E) for more information on ongoing threats to the species.

National and State Legislation

Green turtles in the Central North Pacific DPS are currently protected by the ESA. In Hawaii, they are also protected by the Hawaii Revised Statutes, Chapter 195D (Hawaii State Legislature, accessed 9/10/2010) and Hawaii Administrative Rules, 13-124 (Hawaii Administrative Rules, accessed 9/10/2010), which adopt the same definitions, status designations, and prohibitions as the ESA and carry additional penalties for violations at the State government level. These two statutes have been, and currently are, key tools in efforts to recover and protect this DPS, and both have been effective in improving the status of sea turtles in Hawaii's marine habitat. Additional discussion related to legislation is found below in “Summary of Existing Conservation Efforts”, for example under 14.2.6.2 Federal Laws and Protection (such as establishment of the Papahānaumokuākea Marine National Monument).

The state of Hawaii also has rules associated with coastal zone management (e.g., http://www.capitol.hawaii.gov/hrscurrent/vol04_ch0201-0257/hrs0205a/hrs_0205a-.htm). These rules address establishment of shoreline setbacks and prohibitions relating to mining or taking of sand, as well prohibition of certain structures, unless permitted or authorized by a variance. The state of Hawaii also has a Hawaii Coastal Erosion Management Plan to improve erosion management in the state (http://www6.hawaii.gov/dlnr/ocecl/files/coemap.pdf).

When a species is listed under the ESA, it is automatically listed under the Hawaiian statute. The State is under no obligation to continue to list the species should it be delisted under ESA. If this DPS did not remain listed under the ESA, it is unclear whether it would remain listed under the Hawaiian statute and/or what state protections and management would remain in place.
International Instruments

Many threats to green turtles in the Central North Pacific DPS are addressed, at least in part, by international agreements and conventions, which are listed in Section 14.2.5.6. Hykle (2002) and Tiwari (2002) reviewed the value of some international instruments, which vary in their effectiveness. These instruments can help raise awareness of issues facing sea turtles, promote international collaboration, and sometimes can lead to increased resources (e.g., funding) for sea conservation. However, often, international treaties 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, 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 a 2002 special issue of the Journal of International Wildlife Law and Policy: International Instruments and Marine Turtle Conservation (Hykle, 2002).

14.2.5.5. Factor E: Other Natural or Manmade Factors

Threats, such as incidental bycatch in fishing gear, marine pollution, interactions with recreational and commercial vessels, climate change, beach driving, and major storm events all negatively affect green turtles in the Central North Pacific DPS. Indeed, three of the most common reasons for sea turtle strandings in Hawai'i are entanglement in fishing lines, interactions with fishing hooks, and interaction with marine debris (usually entanglement in nets). Human disturbance (e.g., by tourism) of foraging and basking sea turtles can occur in Hawai'i, however it is unclear what level of threat this disturbance presents.

14.2.5.5.1. Incidental Bycatch in Fishing Gear

Incidental capture in fisheries is a significant threat to green turtles of the Central North Pacific DPS. The primary gear types involved in these interactions include longlines and nets. These are employed by both artisanal and industrial fleets, and target a variety of species.

Longline Fisheries

Pacific longline fisheries capture green turtles as bycatch in longline gear (line, hooks), and these interactions can result in mortality (NMFS, 2012). The Hawai'i-based longline fisheries are expected to kill up to 7 green turtles annually (NMFS, 2012; 2005). Sea turtle bycatch rates in foreign fisheries is estimated to be at least 10 times greater than Hawai'i-based fisheries, perhaps as much as 20 times greater (Bartram and Kaneko, 2004; Kaneko and Bartram, 2008). While exact numbers are not available, it is estimated that, at a minimum, 100 green turtles in the Central North Pacific DPS are captured and killed annually by longline bycatch (NMFS, 2012).

Gill Net Fisheries

Comment [A26]: Comment: So it’s all bad? Nothing good comes of international conventions? Not increased funding for conservation work, not increased awareness, not increased collaboration? This falls far short of being a fair statement... Sorry, I just think that if this is to be raised, do it fairly. Not all is a bad news story 😊

Response: Added language to acknowledge potential positives of the agreements and conventions.

Comment [A27]: Comment: Ok, now is the time to get realistic and determine whether this is a significant threat. Seven. Seven. Just seven. I see that many carcasses on a typical beach... And foreign fleets can not fish in Hawaii, and the turtles don’t leave Hawaii. And genetics tells us the turtles they do catch are not Hawaiian... And when you say 100 turtles, you don’t distinguish by age class, so that 100 juveniles could represent just 10 adults or less. I am ok with listing the numbers, but where above you make it sound like a death knell, here it should be celebrated that all those regulations you have have minimized bycatch to just this, even considering foreign vessels.

Response: This is just stating the facts as we know them. There are turtles captured and killed. Is it "the only" threat, no, but it is one of them (which needs to be considered cumulatively with others).
Interactions between Central North Pacific green turtles and nearshore fisheries in the MHI can result in entanglement, injury, and mortality. Balazs et al. (1987) documented sea turtle mortality resulting from bycatch in fishing gear over 25 years ago in Hawai'i. While gill nets are regulated by the state of Hawai'i, fishers are only required to inspect them completely every two hours, so entanglement and drowning can occur (NMFS, 2012). Each year green sea turtles are incidentally entangled in net gear, some of these resulting in mortality (e.g., Francke et al., 2013), however the reported strandings are believed to be a smaller subset of the actual level of interaction with this gear.

Other Gear Types

Hook-and-line fishing from shore or boats also hooks or entangles green turtles (NMFS, 2012; Francke et al., 2013). Interactions with nearshore recreational fisheries are identified in the NMFS stranding database as those turtles that strand as a result of interactions with fish hooks and fishing line. These include turtles that were hooked externally, ingested hooks, entangled in fishing line, or exhibited intestinal prolapses due to line ingestion. Additionally, net and Gill net entanglement cases include unidentified nearshore and pelagic nets, including cargo nets, trawl nets, lobster nets, and monofilament gill nets. Chaloupka et al. (2008b) reports that between 1982 and 2002 approximately 7 percent of strandings were attributed to hook-and-line fishing gear-induced trauma, and 5 percent for Gill net fishing gear-induced trauma. Nearshore fishery interactions have increased over time with over 60 turtles stranded in 2011 as a result of hook and line interactions, and 46 turtles in 2012 (Francke, 2013; Francke et al., 2013; Ikonomopoulou et al., 2013). While current public outreach efforts by NMFS and its partners are attempting to reduce the magnitude of impact on green turtles from hook-and-line fishing, injury or mortality from the hooking or from the effects of line remaining on turtles that are cut free or break the line remains an issue (http://pifscblog.wordpress.com/2013/06/07/marine-turtle-response-achieves-significant-milestone/).

14.2.5.5.2. Marine Debris and Pollution

The ingestion of and entanglement in marine debris is another anthropogenic threat to Central North Pacific green turtles throughout their range. Green turtles will ingest plastic, monofilament fishing line, and other marine debris (Bjorndal et al., 1994). Effects may be lethal or may be non-lethal but resulting in varying side effects that may increase the probability of death (Balazs, 1985; Carr, 1987; McCauley and Bjorndal, 1999). Impacts of marine pollution can also include contamination from herbicides, pesticides, oil spills, and other chemicals, as well as impacts on water quality (e.g., increases in water column sediments) resulting from structural degradation from excessive boat anchoring, dredging, and other sources (Francour et al., 1999; Lee Long et al., 2000; Waycott et al., 2005).

Kubis and Balazs (2007) describe entanglement and ingestion of marine debris as a potential threat to these turtles, listing discarded or abandoned fishing gear (nets and lines) as well as plastics (bags, 6-pack rings, tar balls, polystyrene or other items that could ensnare or be eaten). Marine debris is common in the MHI and is not only a direct threat to sea turtles, but also to habitat they utilize (Wedding and Friedlander, 2008). Stranding information for this DPS shows that fishing line entanglement is one of the causes of green turtle strandings and mortality in the MHI (Francke et al., 2013; Francke et al., 2014). For example, thirty-six green sea turtles...
stranded in Hawaii in 2012 (Francke et al., 2013) and forty-two in 2013 (Francke et al., 2014) due to line-related entanglement or line ingestion. This number is a subset of the total number of animals possibly affected by this threat (Francke et al., 2013).

In the NWHI, marine debris is also a threat in the terrestrial and marine environment. In 1996, it was estimated that between 750 and 1,000 tons of marine debris were on reefs and beaches in the NWHI, and the sources of much of the debris is fishing nets discarded or lost in the northeastern Pacific Ocean (Keller et al., 2009). This type of debris poses a serious entanglement threat to sea turtles in the NWHI which can result in serious injury or mortality and cause damage to habitat (Wedding et al., 2008). Franke et al. (2013) show that Central North Pacific green turtles can become entangled in net and gill net gear, which can result in mortality. Keller et al. (2009) explain that even if no new debris were to enter the ocean, existing debris in the ocean would continue to accumulate in the NWHI for years. Debris is also a threat to sea turtles in the pelagic zone via ingestion and entanglement, however the extent of impact is much harder to ascertain.

Historic activities in the NWHI have resulted in a legacy of modification and offshore and onshore contamination at FFS, e.g., point sources of Polychlorinated biphenyls (PCBs) due to former Long Range Navigation (LORAN) stations. Elevated levels of contamination remain in soils, nearshore sediment and biota, and pollution (sea and land) related to past and present human activities continues to stress the NWHI ecosystem [Wedding et al., 2008] although we have no evidence that it affects green turtles.

During the 1900s, Johnston Atoll was disturbed by human and military activities such as guano mining, missile launching, airplane operations, nuclear testing and chemical weapons incineration. The lingering effects of these activities include soil contamination (FWS 2013), and turtle foraging habitat at Johnston Atoll appears vulnerable to petroleum contamination (Balazs, 1985). However, the current effect of these activities on the marine environment and sea turtles is unclear.

14.2.5.5.3. Vessel Interactions

As in other parts of the world, boating activities are a threat to turtles within this DPS. To the extent possible, NMFS attempts to determine the causes of strandings in Hawaii. At least 11 green turtles were recorded as having been struck by boats in 2012 alone (Francke et al., 2013). However, given that stranding records are not a complete record of all interactions that may have occurred, these records do not represent the full number of animals struck by boats and numbers are likely higher. Additionally, boat traffic has been shown to exclude green turtles from preferred coastal foraging pastures (Seminoff et al., 2002b), which may negatively affect their nutritional intake.

[Vessel groundings] [mechanical damage to habitat and reef-associated organisms] and related release of contaminants (e.g., fuel, hazardous substances, etc.) are a threat not only to Central North Pacific green turtle habitat, but directly to the animals themselves. This is particularly true in the NWHI, which is exposed to open ocean weather and sea conditions, including severe storm and wave events (Keller et al., 2009). Vessel traffic and presence can also have negative effect through habitat damage from anchors, waste discharge, light and noise (Keller et al., 2009).
14.2.5.4. Climate Change

As in other areas of the world, climate change and sea level rise have the potential to negatively affect green turtles in the Central North Pacific DPS. Global warming and climate change present a number of potential serious threats for green turtles. Climatic considerations such as ocean acidification, temperature changes, and sea level rise could affect feeding ecology, nesting success (via compromising nesting habitat), breeding behavior and timing of nesting, and phenology and spatial distribution of predators (Hawkes et al., 2009). Keller et al. (2009) suggest that sea level rise, changing storm dynamics, sea surface temperatures, and ocean acidification are key threats for the NWHI, and that evidence of sea level rise has already begun to adversely affect terrestrial and ocean habitat. Indeed, Baker et al. (2006) state that climatic changes in the NWHI pose threats through reduction in area of nesting beaches critical to this DPS. Baker et al. (2006) examined the potential effects of sea level rise in the NWHI and found that the primary nesting area for the Central North Pacific population is threatened by sea level rise through possible loss of nesting habitat. Trig, Gin and Little Gin could lose large portions of their area, concentrating nesting even further at East Island (Baker et al., 2006). Additionally, habitat degradation resulting from the release of contaminants contained in landfills and other areas of the NWHI could occur as the islands erode or are flooded from sea level rise (Keller et al., 2009). In contrast to this, Tiwari et al. (2010) argued that East Island itself is still not yet at carrying capacity, in the sense of crude nesting area and current nesting densities. It remains unclear, however, how catastrophic nesting habitat loss and natal homing traits will influence future nesting in this DPS.

Increasing temperatures at nesting beaches may affect hatching development (Chan and Liew, 1995; Godfrey et al., 1996; Marcovaldi et al., 1997; Binckley et al., 1998; Godley et al., 2001; Matsuzawa et al., 2002; Oz et al., 2004; Kaska et al., 2006). For example, changes in temperatures at nesting beaches could affect hatching sex determination and sex ratios (Balazs and Kubis, 2007). Sand temperatures prevailing during the middle third of the incubation period determine the sex of hatching sea turtles and incubation temperatures near the upper end of the tolerable range produce only female hatchlings while incubation temperatures near the lower end of the tolerable range produce only male hatchlings. In addition, as temperatures increase, there is concern that incubation temperatures could reach levels that exceed the thermal tolerance for embryonic development, thus increasing embryo and hatching mortality (Balazs and Kubis, 2007; Fuller et al., 2010a). Niethammer et al. (1997) note that given that the FFS nesting colony is on the northern extreme of green turtle breeding range, small changes in beach conditions (e.g., on microhabitats of nests) may have severe consequences on nesting. Changes in global temperatures could also affect juvenile and adult distribution patterns.

Changes in global temperatures could also affect juvenile and adult distribution patterns. Possible changes to ocean currents and dynamics may also result in negative effects to natural dispersal during a complex life cycle (Van Houtan and Halley, 2011), and possible nest mortality linked to erosion resulting from increased storm frequency (Van Houtan and Bass, 2007) and intensity (Keller et al., 2009).

While sea turtles have survived past eras that have included significant temperature fluctuations, future climate change is expected to happen at unprecedented rates, and if turtles cannot adapt
quickly they may face local to widespread extirpations (Hawkes et al., 2009). Impacts from global climate change induced by human activities are likely to become more apparent in future years (IPCC, 2007).

Beach Driving

Beach driving and other human activities associated with it occur in the MHI, particularly Maui, and are a potential threat to hatchlings that need to reach the water as soon as possible after hatching to avoid certain types of predation. State and Federal agencies are trying to address this threat, but it remains an issue. While beaching driving is not currently affecting many nests or hatchlings, as discussed above, nesting in the MHI is reduced from historical levels, and beach driving remains one among other threats to ensuring nesting increases in the MHI (in order to increase nesting diversity and reduce risks to the DPS).

Major storm events

Natural environmental events, such as cyclones or hurricanes, may affect green turtles in the Central North Pacific DPS. Any significant storm event that may develop could disrupt green turtle nesting activity and hatching production (Van Houtan and Bass, 2007), but would be unlikely to result in whole-scale losses over multiple nesting seasons. However, when combined with the effects of sea level rise, there may be increased cumulative impacts from future storms (Baker et al., 2006).

14.2.6. Summary of Existing Conservation Efforts

There are many ongoing conservation efforts for green turtles in the Central North Pacific DPS by numerous Federal and State agencies and other non-governmental organizations. Intensive monitoring and protective efforts are ongoing in the NWHI, where nesting is occurring in the MHI, and in nearshore waters. While not perfect, regulatory mechanisms in U.S. jurisdiction are in place through the ESA, MSA and the State that currently address direct and incidental take of Central North Pacific green turtles, and these regulatory mechanisms have been an important factor in the encouraging trend in this DPS.

14.2.6.1. State of Hawai‘i

The state of Hawai‘i’s efforts to conserve green turtles include wildlife regulations (discussed in Section 14.2.5.4); coordination of stranding response and specimen storage on Maui, Hawai‘i Island, and Kaua‘i; issuance and management of special activity permits; statewide outreach and education activities; and nest monitoring on Maui (Department of Land and Natural Resources, 2013). It is unclear if sufficient, comprehensive information on fishing distribution and effort in nearshore (state) waters exists and if regulations and implementation are effective.

The Hawai‘i Division of Aquatic Resources (DAR) staff respond to stranded turtle reports and the Division of Conservation and Resources Enforcement investigates reports of illegal poaching, provides support and security at some nest sites and strandings, and addresses
complaints from the public regarding turtle disturbances. DAR also issues special use permits to researchers and educators. Because turtles are already protected under Hawai‘i state law, the current system of Marine Protected Areas (MPAs) throughout the state offers little additional direct benefit but does allow for habitat preservation in certain areas. Through ESA Section 6 (Species Recovery Grant) funding, the Hawai‘i Department of Land and Natural Resources (DLNR) is working cooperatively with NOAA Fisheries Service to minimize certain threats to green turtles in the MHI, such as bycatch in fishing gear and disturbance on beaches.

14.2.6.2. Federal Laws and Protections

Endangered Species Act

The ESA has as its purpose 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. 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 conservation plan (often called HCP for 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. The ESA has been instrumental in curtailing the demise and assisting in the recovery of many species, and green turtles are included among them.

Magnuson-Stevens Fishery Conservation Act

The recently-amended U.S. Magnuson-Stevens Fishery Conservation and Management Act (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 reduction engineering 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."

Papahānaumokuākea Marine National Monument

The Papahānaumokuākea Marine National Monument in the NWHI is a conservation area that encompasses coral reefs, islands and shallow water environments that are important habitats for
rare species such as the threatened green turtle. It was established in June 2006 by Presidential Proclamation, comprising several previously existing federal conservation areas including the NWHI Coral Reef Ecosystem Reserve, Midway Atoll National Wildlife Refuge, Hawaiian Islands National Wildlife Refuge, NWHI Marine Refuge, State Seabird Sanctuary at Kure Atoll and the Battle of Midway National Memorial. The Monument Mission is to carry out seamless integrated management to ensure ecological integrity and achieve strong, long-term protection and perpetuation of NWHI ecosystems, Native Hawaiian culture, and heritage resources for current and future generations. The Monument is administered jointly by three co-trustees – the Department of Commerce through NOAA's Office of National Marine Sanctuaries and NMFS Pacific Islands Regional Office; Department of the Interior through the FWS’s Pacific Region National Wildlife Refuge System and Pacific Islands Fish and Wildlife Office; and the State of Hawai’i through the DLNR’s’ DAR and Division of Forestry and Wildlife. The Monument is working to reduce threats through an ecosystem approach to management. This includes the development of an effective regulatory framework and permitting process, education and outreach, preventative measures to minimize risk, and response and restoration to damaged or degraded natural resources.

**Johnston Atoll**

The Pacific Remote Islands Marine National Monument was established in January 2009, and is cooperatively managed by the Secretary of Commerce (NOAA) the Secretary of the Interior (FWS), with the exception of Wake Island and Johnston Atoll which are currently managed by the Department of Defense. The areas extend 50 nautical miles from the mean low water lines and include green turtle habitat. The protected area provides some protection to sea turtles and their habitat (through permitted access) and its remoteness.

**Fishing Around Sea Turtles Outreach (FAST)**

To raise awareness among fishers to reduce impacts to sea turtles around the Main Hawaiian Islands, NMFS has developed a "Fishing Around Sea Turtles" (FAST) program to promote "Turtle Friendly" fishing gear, such as barbless circle hooks, and provide best-practice guidelines to assist hooked or entangled turtles so fishermen can support the recovery of sea turtles in Hawai’i. The program also includes practical fishing tips suggested by fishermen that may reduce the potential for interactions, and encourages reporting injured or dead turtles to NMFS’ sea turtle stranding program. FAST was developed in 2010 (and refined 2012) through a multi-agency partnership that includes NMFS, the State of Hawai’i, the Western Pacific Fisheries Management Council, local experts, and fishers. The effects of this program on population dynamics is not yet known, however it is hoped the program will help ameliorate effects of near shore fishing.

14.2.6.3. **Non-governmental and Multi-agency Efforts**

Numerous non-governmental organizations assist in the conservation of Hawai’i’s green turtles, either by conducting public outreach programs, protecting basking green turtles, conducting beach monitoring of turtles, and/or conducting in-water surveys. These organizations include, but are not limited, to the following: The Nature Conservancy, Conservation Council for
Hawai'i, Malama na Honu, Hawai'i Wildlife Fund, Na Kama Kai, Malama Waimea-Pupukea, Waikiki Aquarium, Sea Life Park, and Maui Ocean Center. Other groups that promote NOAA viewing guidelines and conduct outreach to minimize disturbance of sea turtles include the Coral Reef Alliance (CORAL), Reefwatch, Aston and Sheraton Hotels, Roberts Hawai'i, Snorkel Bob, Pacific Whale Foundation, Trilogy Excursions, Adventure Cruises, and Explorer.

Sea Life Park Hawai'i conducts public educational programs via green turtles on display, and has been a partner in developing a technique that uses microchips in hatchling green turtles. In addition, although the conservation value is unclear, Sea Life Park has tagged and released approximately 200 juvenile green turtles (2-8 years of age), and over 13,000 hatchlings (G. Balazs, NMFS, pers. comm., 2013).

Debris “clean up” efforts are also conducted in Hawai'i by both the NOAA Marine Debris Program and non-government organizations (Friedlander et al., 2008). These efforts are useful, but debris cleanup is a continuing challenge.

14.2.6.4. International Instruments

Several regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the Central North Pacific. The international instruments listed below apply to sea turtles found in this area and are described in Appendix 5.

- Convention on Biological Diversity
- Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention)
- Convention on the Conservation of Migratory Species of Wild Animals
- Convention on International Trade in Endangered Species of Wild Fauna and Flora
- Convention for the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean
- Fishery and Agricultural Organization Technical Consultation on sea turtle-fishery interactions
- Indian Ocean-South-East Asian Marine Turtle Memorandum of Understanding
- Inter-American Convention for the Protection and Conservation of Sea Turtles
- International Convention for the Prevention of Pollution from Ships (MARPOL)
- International Union for Conservation of Nature
- Ramsar Convention on Wetlands
- Secretariat of the Pacific Regional Environment Programme
- United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing
- United States Magnuson-Stevens Conservation and Management Act

14.3. Assessment of Significant Portion of its Range (SPR)
The SRT reviewed the information on threats and extinction risk within each DPS to determine whether there were portions of the range of the DPS that might be considered “significant portions of the range” in accordance with the definition of endangered and threatened species and the draft policy that interprets that term (see Section 3.4 for more details on the SPR deliberative process).

For the Central North Pacific DPS, 96 percent of all nesting occurs at one nesting site (FFS) with some nesting occurring at other sites in the NWHI and in the MHI Islands within the DPS. This level of concentration of nesting far exceeds that of any other DPS. Recent documentation of small numbers of nesters at various sites in the main Hawaiian Islands is encouraging because, although these nesters represent a small portion of overall nesting, they play a significant role as potential refugia nesting populations should some nesting at FFS be lost.

Threats are not uniformly distributed across the DPS. There is a long-term threat of loss of nesting area due to sea level rise at FFS and other sites in the NWHI, although the implications of this loss are not fully understood. There are greater in-water threats in the nearshore areas of the MHI and potentially greater threats of human disturbance to nests in the MHI. Most importantly, nests in the MHI are at inherently high extinction risk due to small-population effects.

The SRT concluded that, because of disparities in risk, SPR considerations might apply to this DPS. Although FFS might be at greater long-term risk of extinction from sea-level rise, currently it supports that vast majority of the nesting within the DPS. Collectively the many small sites on the MHI used recently or intermittently for nesting appear to be at higher current risk due to small-population effects. Therefore, the SRT concluded that the nesting sites outside of FFS might constitute a significant portion of the DPS’ range.

Following the procedure outlined in the draft interagency policy on SPR, therefore, the next step in this analysis was to determine whether, if all of the small populations outside of FFS were lost, extinction risk of the population at FFS would be substantially increased. If so, the portion of range outside FFS would be considered “significant” under the ESA.

The next step in this analysis was to determine whether, if all of the small populations outside of FFS were lost, extinction risk of the population at FFS would be substantially increased.

14.4. Assessment of Extinction Risk
Factors that affect population dynamics (and potentially extinction risk) have a certain level of uncertainty associated with them. For example, populations can be affected by “environmental uncertainty” (random or unpredictable changes in food supply, predators, parasites, etc.); another example is natural catastrophes (Shaffer, 1987; Hunter and Gibbs, 2007). These kinds of risks become particularly important for smaller, “single,” and concentrated populations that lack diversity. The current Central North Pacific nesting population represents a small, essentially single population, concentrated primarily at one location. While more than 96 percent of nesting occurs at one site, the nesting population trajectory is positive and encouraging. The PVA analyses for the East Island time series shows a 0.0 percent probability that the population will drop below either the trend or the absolute abundance thresholds. However, the characteristics of this nesting population make the population highly vulnerable to environmental uncertainty and natural catastrophes, significantly increasing the probability and risk of extinction of the overall Central North Pacific population, even though the population may have positive population growth.

For the SRT’s assessment of extinction risk for green turtles in the Central North Pacific DPS, there were two separate sets of ranking exercises: one focusing on the importance that each SRT member placed on each of the six different elements for this region (Table 14.3), and a second which reflects the SRT members’ expert opinion about the probability that green turtles would fall into any one of various extinction probability ranges (Table 14.4). See Section 3.3. for details on the six elements and the voting process. Both of these exercises had to be completed twice, once for the entire DPS, and once for the DPS assuming an SPR was extirpated and only the nesting populations of currently stable or increasing beaches remained (see Section 3.4 for more discussion of Significant Portion of its Range (SPR)).

14.4.1. Risk Assessment Voting For Entire DPS

The SRT first conducted voting on both the six elements and the overall risk of extinction for the entire DPS.

Table 14.3. Summary of ranks that reflect the importance placed by each SRT member on the critical assessment elements considered for the entire Central North Pacific DPS. For Elements 1-4, higher ranks indicate higher risk factors.
With respect to the important rankings for the six critical assessment, the first four elements using the 1-5 ranking system (higher rank equals higher risk factor), Spatial Structure featured most prominently in the risk threshold voting, most likely because 96 percent of the nesting activity occurs at one location in the Central North Pacific DPS. Nesting Abundance and Diversity/Resilience also featured significantly in the risk threshold voting. SRT members weighed future threats not yet experienced by the population to be more significant than emerging conservation efforts in their risk assessment voting. With respect to the diversity of opinions among the SRT members when considering the six Critical Elements, the Diversity / Resilience element had the largest range of 1 to 5, the largest possible range. The Spatial Structure and Abundance elements also had large ranges of 1 to 4. This spread of values reflects the SRT members attributing different levels of significance to the concentration of nesting almost entirely at one location in this DPS.

Table 14.4. Summary of Green Turtle SRT member expert opinion about the probability that the Central North Pacific DPS will reach a critical risk threshold within the next 100 years throughout all of its range. Each SRT member is assigned 100 points across the rank categories. The continuum in Row 1 has categories with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability Of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1-5%</th>
<th>6-10%</th>
<th>11-20%</th>
<th>21-50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN ASSIGNED POINTS</td>
<td>49.1</td>
<td>16.3</td>
<td>14.9</td>
<td>7.5</td>
<td>7.3</td>
<td>4.9</td>
</tr>
<tr>
<td>SEM</td>
<td>11.6</td>
<td>3.8</td>
<td>4.5</td>
<td>3.3</td>
<td>4.4</td>
<td>3.5</td>
</tr>
<tr>
<td>Min</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Max</td>
<td>98</td>
<td>40</td>
<td>50</td>
<td>35</td>
<td>52</td>
<td>39</td>
</tr>
</tbody>
</table>

Of the six critical risk threshold categories describing the probability that the Central North Pacific DPS will reach a critical risk threshold within 100 years (Table 14.4), the SRT member votes resulted in the greatest point (i.e., probability) designation in the '<1 %' risk range, with a mean of 49.1 points. The categories with the fewest allocated points were the '>50 %' and '21-50 %' ranges, with means of 4.9 and 7.3, respectively.

Nearly half of the points indicated relatively low risk, and explanations provided included the nesting time series growth trend and PVA results. To a lesser extent, members recognized that conservation and enforcement in this DPS helped its future trajectory. Reasons for higher risk scores included the high concentration of nesting at one site and low nesting abundance in this DPS as the primary concerns. In their vote justifications, most members cited the high concentration of nesting at one site and low nesting abundance in this DPS as the primary concerns influencing their vote. Additional concerns that were cited included the significant historical reduction in nesting areas, the lack of current diversity in nesting areas, and the impacts of climate change. Conversely, many members were encouraged by the nesting time

Comment [A45]: Comment: And yet in your text earlier you note that it is not possible to determine what these future threats would be so they were not considered. Is this not a reversal of the way the process was considered?

Response: Added text to clarify.

Comment [A46]: Response: Added text to clarify.

Comment [A47]: Comment: These all seem to contradict the findings, whereby most felt the risk to extinction was <1%. Would the arguments not be along the lines of ‘good legal protection’, ‘isolated protected nesting site’, ‘increasing populations’ etc etc. If the results were like this why defend them with negative comments? Irrespective of the second half of the paragraph....

Response: Added text to clarify. Additionally, most did not feel the risk was <1%. The mean assigned points votes for <1% was 49.1. Over 50 of the mean assigned points were >1%, some as high as 21-50% and 4.9 were >50%. This is not contradictory, just presents what the voters voted and some of the reasons behind it (both concerns and encouraging aspects). The members did recognize the encouraging increasing population, but also recognized the concerns of high concentration of nesting at one site, low nesting abundance, etc.
series growth trend and PVA results. To a lesser extent, members recognized that conservation
and enforcement in this DPS helped its future trajectory. Reasons for higher risk scores included
the high concentration of nesting at one site and low nesting abundance in this DPS as the
primary concerns. Additional concerns that were cited included the significant historical
reduction in nesting areas, the lack of current diversity in nesting areas, and the impacts of
climate change. The vote justifications provided for this DPS exhibited differences of opinions
across SRT members, depending on which factors each weighed as most significant. Due to these
conflicting factors, the vote justifications provided for this DPS were somewhat inconsistent
across SRT members, depending on which factors they weighed as most significant.

14.4.2. Extinction Risk with SPR Consideration

Because the SRT determined that an SPR potentially exists within this DPS, the SRT also had to
repeat the voting on both the six elements and the overall risk of extinction, assuming that the
SPR (MHI population) was lost. See Section 3.3. for details on the six elements and the voting
process.

Table 14.5. Summary of ranks that reflect the importance placed by each SRT member on the
critical assessment elements considered for the Central North Pacific DPS, assuming the SPR is
lost. For Elements 1–4, higher ranks indicate higher risk.

<table>
<thead>
<tr>
<th>Critical Assessment Elements</th>
<th>Abundance (1 to 5)</th>
<th>Trends / Productivity (1 to 5)</th>
<th>Spatial Structure (1 to 5)</th>
<th>Diversity / Resilience (1 to 5)</th>
<th>Five-Factor Analyses (-2 to 0)</th>
<th>Conservation Efforts (0 to 2)</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN RANK</td>
<td>2.67</td>
<td>1.33</td>
<td>3.25</td>
<td>2.67</td>
<td>-0.92</td>
<td>0.58</td>
</tr>
<tr>
<td>SEM</td>
<td>0.28</td>
<td>0.19</td>
<td>0.37</td>
<td>0.45</td>
<td>0.23</td>
<td>0.19</td>
</tr>
<tr>
<td>RANGE</td>
<td>1–4</td>
<td>1–3</td>
<td>1–5</td>
<td>1–5</td>
<td>(-2) – 0</td>
<td>0–2</td>
</tr>
</tbody>
</table>

With respect to the important rankings for the six critical assessment elements (Table 14.5),
Spatial Structure again featured most prominently in the risk threshold, although this time with
an even higher score (indicating higher risk), no doubt because if the MHI nesters were no longer
extant, 100 percent, as opposed to 98 percent of the nesting activity would occur at one location
in the Central North Pacific DPS (NWHI). Nesting Abundance and Diversity/ Resilience again
featured significantly in the risk threshold voting, although were virtually unchanged (2.67 vs.
2.7). Future threats not yet experienced by the population was also virtually unchanged (-0.92
vs. -0.9) and remained nearly twice as significant as emerging conservation efforts in their risk
assessment voting, with conservation efforts also being virtually unchanged (0.58 vs. 0.6). With
respect to the diversity of opinions among the SRT members when considering the six Critical
Elements, the Diversity/ Resilience element remained unchanged with the largest range of 1 to 5,
the largest possible range. The range in the Spatial Structure element increased to a range of 1 to
5 (as opposed to the previous range of 1 to 4), no doubt attributable to an even more constricted
range. The spread of values reflects the SRT members attributing different levels of significance
to the concentration of nesting almost entirely at one location in this DPS.
Table 14.6. Summary of Green Turtle SRT member expert opinion about the probability that the Central North Pacific DPS will reach a critical risk threshold within the next 100 years, without the SPR. Each SRT member is assigned 100 points across all rank categories. The continuum in Row 1 has categories with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1-5%</th>
<th>6-10%</th>
<th>11-20%</th>
<th>21-50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN ASSIGNED POINTS</td>
<td>46.25</td>
<td>17.17</td>
<td>15.50</td>
<td>7.92</td>
<td>7.83</td>
<td>5.33</td>
</tr>
<tr>
<td>SEM</td>
<td>11.76</td>
<td>4.06</td>
<td>4.62</td>
<td>3.38</td>
<td>4.16</td>
<td>3.89</td>
</tr>
<tr>
<td>Min</td>
<td>0</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Max</td>
<td>98</td>
<td>40</td>
<td>50</td>
<td>35</td>
<td>49</td>
<td>44</td>
</tr>
</tbody>
</table>

With respect to the overall risk of extinction, or the six critical risk threshold categories describing the probability that the Central North Pacific DPS will reach a critical risk threshold within 100 years (Table 14.6), the SRT member votes again resulted in the greatest point (i.e., probability) designation in the '<1 %' risk range, although this time with a mean of 46.25 rather than 49.1 points. The categories with the fewest allocated points were the '>50 %' and '21-50 %' ranges, with means of 5.33 and 7.83 respectively (compared with 4.9 and 7.3, respectively for the entire DPS).

The combined expert judgment of the SRT is that the DPS would be at a slightly increased risk of extinction if the SPR was lost, with 46 vs. 49 percent chance that the population has a <1 percent risk of extinction, and a 54 percent vs. 51 percent chance that the population has >1 percent risk of extinction. This appears to be due to increases in risk due to decreased spatial structure.

### 14.5. Synthesis and Integration

During the analysis of the status of the Central North Pacific DPS, an integrated approach was taken by the SRT to consider the many critical assessment elements described earlier. The Central North Pacific DPS is characterized by geographically concentrated nesting and moderately low levels of abundance. Such a low number is the result of chronic historical exploitation which extirpated significant nesting grounds. Nesting has been documented recently in 12 different locations. Sea level rise and other climate change impacts are a concern because more than 98 percent of nesting occurs in remote low-lying oceanic atolls. However, scientific monitoring, conservation efforts, and legal enforcement are all very good and favor the persistence of this DPS. Moreover, time series analysis of nester abundance over 40 years at the major nesting site is encouraging, showing an annual rate of increase of 4.8 percent. Results from the PVA indicate a 0.0 percent probability the population will fall below either reference point in the next century, which is the lowest possible risk, although we noted that the PVA modeling used here has not accounted for all considerations.

Comment [A52]: Comment: Not to mention there is a lot more area available to them if they need it. And they have a record of making use of this....

Responses: Noted and mentioned in the text above.
Votes on both the importance placed on critical elements and extinction risk varied widely among SRT members. Overall, Justifications of votes by members were mixed on this DPS. Members attributed the largest probability (49.8) to the lowest single category of extinction risk (<1 percent). This is likely due to the 40 year nesting trends at the largest nesting site and the PVA forecast results. However, the unprecedented concentration, the moderately low abundance of nesting and the threats from climate change likely accounted for a mean of 20 percent probability of having at least an 11 percent extinction risk. As stated earlier, the vote justifications provided for this DPS were somewhat inconsistent across SRT members, depending on which factors they weighed as most significant.

However, justifications of votes by members indicated that the unprecedented concentration of nesting at one site, the moderately low abundance of nesting and the threats from climate change likely accounted for the wide spread of points in higher risk categories (Figure 14.5). As stated earlier, the vote justifications provided for this DPS differed widely across SRT members, depending on which factors they weighed as most significant.

It is important to note that our analysis did not consider the scenario in which current laws or regulatory mechanisms were not continued. For instance, if the protections of the ESA were no longer in place for this DPS, both the on-the-ground conservation actions as well as financial and other resources that were afforded by the ESA, may not continue at the same level.
15. **EAST PACIFIC DPS (DPS #11)**

15.1. **DPS Range and Nesting Distribution**

The East Pacific DPS extends from the California/Oregon boarder, USA (41˚N) southward along the Pacific coast of the Americas to central Chile (40˚S). The northern and southern boundaries of this DPS extend from the aforementioned locations in US and Chile to 222˚W and 264˚W, respectively. The offshore boundary of this DPS is a straight line between these two coordinates. This DPS encompasses the Revillagigedos Archipelago (Mexico) and Galapagos Archipelago (Ecuador).

Green turtle nesting is widely dispersed in the Eastern Pacific Ocean (Figure 15.1). The two largest nesting aggregations are found in Michoacán, Mexico and in the Galapagos Islands, Ecuador (Zárate *et al*., 2003; Delgado-Trejo and Alvarado-Figueroa, 2012). Secondary nesting areas are found throughout the Pacific Coast of Costa Rica and Clarion and Socorro Islands in the Revillagigedos Archipelago, Mexico. Low level nesting occurs in Colombia, Ecuador, Guatemala, and Peru. Scattered nesting also occurs from Mexico's Baja California Peninsula (G. Tiburcios-Pintos, Minicipio de Los Cabos, pers. comm., 2012) to Peru (S. Kelez, ecOceanica, pers. comm., 2012; Figure 15.1).

![Figure 15.1](image)

**Figure 15.1.** Nesting distribution of green turtles in the East Pacific DPS (blue-shaded area marked with ‘11’). Size of circles indicates nesting abundance category. Locations marked with ‘×’ indicate nesting sites lacking abundance information.
15.2. Critical Assessment Elements

In the evaluation of extinction risk for green turtles in the East Indian-West Pacific DPS, the SRT considered six distinct elements, each of which are discussed below: (1) Nesting Abundance, (2) Population Trends (3) Spatial Structure, (4) Diversity / Resilience, (5) Five-Factor Threat Analyses, and (6) Existing Conservation Efforts. See Section 3.3 for additional information on the selection of these six critical assessment elements.

15.2.1. Nesting Abundance

For the East Pacific DPS, we identified 36 total nesting sites for which abundance information is available. There are sporadic nesting events in many other areas in the East Pacific DPS, but nesting abundance is undocumented and therefore we do not report these sites here (but see x’s in Figure 15.1). Of these sites, there are two primary nesting concentrations (Michoacán, Mexico, and the Galapagos Islands, Ecuador) and a complex of beaches in Costa Rica that, although lesser in magnitude than Mexico and Galapagos, bares mention due to the apparently large numbers of green turtles that nest each year (M. Heidermeyer, Univ. Costa Rica, pers. comm., 2013). In this context we summarize nesting activity in the three countries below:

Mexico

Based on these nesting beach monitoring efforts, it is apparent that the current adult female nester population for Colola, Michoacán is 11,588 females, which makes this the largest nesting aggregation in the East Pacific DPS, comprising nearly 58 percent of the total adult female population.

The highest nesting densities for the state of Michoacán are at Colola and Maruata Beaches. The longest-term data available are for Colola, where nesting beach monitoring has been ongoing every year since the 1981–1982 nesting season. This site accounts for ~74.4 percent of green turtle nesting in the State of Michoacán; Maruata contains 24.1 percent of the nesting within the state (Delgado and Alvarado-Díaz, 2006; C. Delgado, Universidad Michoacana, pers. comm., 2007). Nesting in Michoacán has been quantified at three additional beaches (Llorona, Motín de Oro, Aguas Blancas), but occurs throughout the state.

The Revillagigedos Islands are a secondary nesting site. There are three areas where green turtles nest: Academy Bay and Playas Blancas on Clarion Island (Brattstrom, 1982; Awbrey et al., 1984), and Sulfur Bay on Socorro Island (Márquez-Millán, 1990). From 1999–2001, a mean of 47 nests were deposited each year at Socorro Island, and a mean of 79 nests were deposited each year at Clarion Island (Juárez-Ceron et al., 2003). However, during a survey in 2008 on Clarion Island, Holroyd and Telfry (2010) quantified body bits and estimated that as many as 500 green turtle nests were laid over a 4-week period. However, as with all single year data sets, especially those based on counts of body pits, the data should be viewed with caution.
Ecuador

In the Galapagos Islands (Ecuador), nesting at the four primary nesting sites (Quinta Playa and Barahona-Isabela Island, Las Bachas-Santa Cruz Island, and Las Salinas - Baltras Island) has been stable to slightly increasing since the late 1970s. Mean annual nesting abundance at these sites was 1,283 females from 1979 to 1980 to from 1982 to 1983 (Green and Ortiz-Crespo, 1982; M. Hurtado, Hurtado and Associates, Inc., unpubl. data). From 2001 to 2002 to from 2005 to 2006, a total of 3,603 adult female nesters in the population (Zárate et al., 2006). Based on these data, it is apparent that the Galapagos nesting concentration is currently the second largest nesting assemblage for green turtles in the eastern Pacific Ocean, following only that of Michoacán, which has a total of 11,588 nesters (Delgado-Trejo and Alvarado-Figueroa, 2012).

Costa Rica

Green turtles nest throughout Costa Rica and we have identified at least 26 nesting sites that together host upwards of 2,800 nesting females in the population. The most significant green turtle nesting aggregations in Costa Rican territories are found along the northern Pacific coast, along the Nicoya Peninsula, which is divided into the Guanacaste Conservation Area and the Tempisque Conservation Area. In both Conservation Areas, nesting beaches are typically from 0.5 to 2 km long. Cabuyal Beach and Nombre de Jesús Beach may host up to 273 and 450 nesters, respectively (P. Santidrián-Tomillo, The Leatherback Trust, pers. comm., 2012); numerous other beaches host ≤10–50 nests/yr.

Table 15.1. Summary of green turtle nesting sites in the East Pacific DPS. Data are organized by country, nesting site, monitoring period, and estimated total nester abundance [(Total Counted Females/Years of Monitoring) x Remigration Interval], and represent only those sites for which there is sufficient data to estimate abundance. Remigration interval for green turtles in the East Pacific is calculated at 3yrs (Alvarado-Díaz and Figueroa 1990). For a list of references for these data, see Appendix 2.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Mexico</td>
<td>Colola, Michoacán</td>
<td>2010–12</td>
<td>11,588</td>
</tr>
<tr>
<td>Mexico</td>
<td>Llorona, Michoacán</td>
<td>2007</td>
<td>90</td>
</tr>
<tr>
<td>Mexico</td>
<td>Bahia Maruata, Michoacán</td>
<td>2007</td>
<td>1,149</td>
</tr>
<tr>
<td>Mexico</td>
<td>Motín de Oro, Michoacán</td>
<td>2007</td>
<td>240</td>
</tr>
<tr>
<td>Mexico</td>
<td>Arenas Blancas, Michoacán</td>
<td>2007</td>
<td>90</td>
</tr>
<tr>
<td>Mexico</td>
<td>Cape Region, BCS</td>
<td>2007–09</td>
<td>7</td>
</tr>
<tr>
<td>Mexico</td>
<td>Revillagigedos, MX</td>
<td>November-December 2008</td>
<td>500</td>
</tr>
</tbody>
</table>

Comment [A4]: Numbers for the Nombre de Jesus complex seems underestimated. Additionally, there are other nesting beaches in this area that register a higher number of turtles but where monitoring programs have only started recently (i.e. Islas Muercielagos). There are other beaches, such as Prieta, Virador and Matapalo, that could have similar levels than Cabuyal. Baulas Park gets ~ 20-30 nests per year (< 10 females).

Response: WE AGREE THAT THE NUMBERS COULD BE HIGHER. HOWEVER, WITHOUT DATA WE ARE UNABLE TO SPECULATE ON THIS POSSIBILITY.

Comment [A5]: What remigration interval was considered?

Response: revised text
<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Costa Rica</td>
<td>Playa Junquillal</td>
<td>June 2012-March 2013</td>
<td>48</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa San José, Bat Islands</td>
<td>November-March 2013</td>
<td>498</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Colorada</td>
<td>4 observations, 2 in January 2013, 2 in March 2013</td>
<td>498</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Nancite</td>
<td>2012</td>
<td>123</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Naranjo</td>
<td>2012</td>
<td>48</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Cabuyal</td>
<td>2012</td>
<td>273</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Prieta</td>
<td>1 time observation January 2010</td>
<td>75</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Virador</td>
<td>1 time observation January 2010</td>
<td>75</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Matapalo</td>
<td>1 time observation January 2010</td>
<td>75</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Blanca</td>
<td>1 time observation January 2010</td>
<td>75</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Nombre Jesús</td>
<td>2012</td>
<td>450</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Grande /Ventanas</td>
<td>Year round</td>
<td>48</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Langosta</td>
<td>November-March 2013</td>
<td>48</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Avellanas</td>
<td>October-March 2012</td>
<td>9</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Lagartillo</td>
<td>October-March 2012</td>
<td>48</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Callejones</td>
<td>October-March 2012</td>
<td>48</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Blanca</td>
<td>October-March 2012</td>
<td>48</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Junquillal</td>
<td>Year round</td>
<td>48</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Ostional</td>
<td>October-April 2012</td>
<td>48</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Buena Vista</td>
<td>July-December 2009</td>
<td>9</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Camaronal</td>
<td>2012</td>
<td>48</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Corozalito</td>
<td>June-December 2012</td>
<td>9</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa San Miguel</td>
<td>2012</td>
<td>9</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Playa Caletas</td>
<td>June-December 2012</td>
<td>9</td>
</tr>
<tr>
<td>Costa Rica</td>
<td>Punta Banco</td>
<td>June-December 2010</td>
<td>9</td>
</tr>
<tr>
<td>Colombia</td>
<td>Isla Gorgona</td>
<td>2007–2009</td>
<td>4</td>
</tr>
<tr>
<td>Ecuador</td>
<td>Galapagos</td>
<td>2003–2005</td>
<td>3,603</td>
</tr>
</tbody>
</table>
Table 15.2. Green turtle nester abundance distribution in the East Pacific DPS.

<table>
<thead>
<tr>
<th>COUNTRY</th>
<th>NESTING SITE</th>
<th>MONITORING PERIOD (YRS)</th>
<th>ESTIMATED NESTER ABUNDANCE</th>
</tr>
</thead>
<tbody>
<tr>
<td>Ecuador</td>
<td>Mainland</td>
<td>2010</td>
<td>15</td>
</tr>
</tbody>
</table>

15.2.2. Population Trends

There are two sites in the Eastern Pacific for which some level of knowledge is available regarding nesting trends, only one of which—Colola, Mexico—met the qualifications for reporting nesting trends; in this case there were enough data of high quality to conduct population viability analyses (a minimum of 15 years of recent nesting data with an annual nesting level of more than 10 females; for more on data quantity and quality standards used, see Section 3.2). For a list of references on trend data, see Appendix 3.

To assist in interpreting these PVAs, we indicate the probability of green turtle nesting populations declining to two different biological reference points, one using a trend-based and the other an abundance-based threshold. The trend-based reference point for evaluating population forecasts is half of the last observed abundance value, i.e., a 50 percent decline. The abundance-based reference point was a total adult female abundance of 300 females. Risk is calculated as the percentage of model runs that fall below these reference points within 100 years. For a full discussion of these PVAs and these reference points, see Section 3.2. Population viability analysis indicates that the population will likely continue to increase.

Comment [A7]: Total nesting population or annual levels?
Response: TOTAL FEMALE NESTING LEVELS.
Figure 15.2. Stochastic Exponential Growth (SEG) Model Output for Colola, Michoacan, Mexico. Black line is observed data, dark green line is the average of 10000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference. Nesters were computed from nest counts using 3.1 nests per female (Alvarado-Díaz et al. 2003).

The PVA indicates that there is a 4.9 percent probability that this population will fall below the trend reference point (50 percent decline) at the end of 100 years. The PVA also indicates a 0.3 percent probability that this population falls below the absolute abundance reference (100 females per year) at the end of 100 years. This PVA has important limitations, and does not fully incorporate other key elements critical to the decision making process such as spatial structure or threats. It assumes all environmental and anthropogenic pressures will remain constant in the forecast period and it relies on nesting data alone. For a full discussion of these PVAs and these reference points, see Section 3.2.

This observed increase may have resulted from the onset of nesting beach protection in 1979—as is suggested by the similarity in timing between the onset of beach conservation and the age-to-maturity for green turtles in Pacific Mexico. The initial upward turn in annual nesting was seen in 1996, about 17 years after the initiation of a nesting beach protection program (Clifton et al., 1982; Alvarado-Díaz et al., 2001), and growth data from the Gulf of California suggest that green turtles mature at about 15–25 years (Seminoff et al., 2002). Although not a clear cause of the increasing nesting trend, the consistency in timing is nonetheless compelling. The presidential decree protecting all sea turtles of Mexico (Pesca, 1990) certainly helped the situation, but this occurred much later than the start of nesting beach conservation. It is more likely that this national legislation has had its greatest positive impact at the foraging areas, where green turtle hunting was once rampant.
15.2.3. Spatial Structure

When examining spatial structure for the Eastern Pacific DPS, the SRT examined three lines of evidence, including genetic data, flipper and satellite tagging, and demographic data.

Genetic sampling in the eastern Pacific has been extensive and the coverage in this region is substantial considering the relative low population sizes of most eastern Pacific rookeries. Within this DPS there is significant population substructuring (pairwise FST 0.08–0.29, p<0.005). Four regional genetic stocks have been identified in the eastern Pacific (P. Dutton, NMFS, unpubl. data): Revillagigedos Archipelago (Mexico), Michoacán (Mexico), Central America (Costa Rica) and the Galapagos Islands.

There is a relatively high level of spatial structure and the presence of rare/unique haplotypes at each nesting site stock. Green turtles from multiple nesting beach origins commonly mix at feeding areas in the Gulf of California (Nichols, 2003; P. Dutton, NMFS, unpubl. data). Along the Pacific coast and in San Diego Bay (USA), the existing haplotype frequencies of foraging turtles suggest that these sites have substantially greater input from the Revillagigedos Islands than from Michoacán (Nichols, 2003; P. Dutton, NMFS, unpubl. data). Green turtles foraging at Gorgona Island in Colombia showed that most (>80 percent) of the turtles originated from rookeries in the Galapagos Islands. They also found a small contribution from Michoacán, Mexico (Amorocho et al., 2012). There is a rare occurrence (<5 percent) of turtles with the haplotype discovered to be common in nesting green turtles from the Central West Pacific Ocean (Amorocho et al., 2012; P. Dutton, NMFS, unpubl. data). There are very rare occurrences of green turtles with eastern Pacific origins in Hawaiian (Dutton et al., 2008), and Japanese waters (Kuroyanagi et al., 1999; Hamabata et al., 2009), and as bycatch in fisheries operating in the North Central Pacific Ocean (Parker et al., 2011). A recent study using nuclear SNPs and microsatellite markers investigated the genetic stock structure among five Pacific green turtle nesting populations. They found significant structure between their two eastern Pacific sample sites (Galapagos and Mexico; FST=0.02, p<0.001) suggesting that male-mediated gene flow between regional nesting stocks is limited (Roden et al., 2013).

Flipper tag recoveries show 94 tag returns from foraging areas that were applied at two primary nesting sites, Michoacán Mexico and the Galapagos Islands, Ecuador. Two apparent groupings suggest some North/South structure. There are 49 satellite tracks of green turtles in the eastern Pacific show apparent track clustering in Northwest Mexico to Southern United States, and in the Southeast Pacific, from the Galapagos Islands to the high seas, and also to the Central American Mainland. The number of satellite tracks are too few to provide solid information on spatial structure.

The primary demographic features of green turtles that are relevant for interpreting population structure and long term trends include age-to-maturity (often via growth studies), reproductive longevity, sex ratio, reproductive output (i.e., egg production, clutch frequency, hatching success, inter-nesting interval), and annual survivorship. Seminoff et al. (2002) reports 9–21 years to reach sexual maturity after settling into this neritic foraging areas on northwest Mexico. However, a study in San Diego Bay, USA, found very high growth rates (McDonald Dutton and Dutton, 1998; Eguchi et al., 2012).
Within region variation for any one of these components may suggest a level of spatial structure for the East Pacific DPS. Among all nesting assemblages in the East Pacific DPS, the Revillagigedos Islands stands out as uniquely different from the remaining areas. Females nesting in Michoacán are substantially smaller than those nesting in the Revillagigedos (82 cm vs. 94 cm mean CCL; Alvarado-Díaz and Figueroa, 1990; Juarez-Ceron et al., 2003). The estimated Age to Maturity is 9–47 years. In-water survivorship is known for very few areas. Survivorship tends to be lower for juveniles and subadults (0.58) than for adult green turtles (0.97) in northwest Mexico (Seminoff et al., 2003). A study in a northern foraging area in the U.S. (San Diego Bay) indicated an average annual survival rate of 0.86 (95 percent CI=0.36–0.99), which included a wide range of age groups (Eguchi et al., 2012).

15.2.4. Diversity/Resilience

Within the eastern Pacific Ocean, specific or subspecific status has been applied to green turtles (also known as black turtles; C. mydas agassizii) ranging from Baja California south to Peru and west to the Revillagigedos Islands and Galápagos Archipelago (Pritchard, 1997; Marquez-Millan, 2007); however, genetic analyses do not support such taxonomic distinctiveness (Bowen et al., 1992; Karl et al., 1992). Moreover, this genetic information is more germane to the overall global genetic diversity of green turtles (Section 4); there is no genetic information that suggests green turtles in the East Pacific DPS should be split into separate subspecies. The aspects considered under this critical assessment element include the overall nesting spatial range, diversity in nesting season, diversity of nesting site structure and orientation (e.g., high vs. low beach face, insular vs. continental nesting sites), and the genetic diversity within the DPS. Aspects such as these are important considerations for assessing the potential impact of catastrophic events such as storms, sea level rise, and disease.

The East Pacific DPS has substantial nesting at both insular and continental nesting sites. There are varying levels of threats in each of these sites (Zárate, 2012). Insular sites have very low levels of human interference at nesting beaches, although animals may be impacted in foraging areas. The low impacts at insular nesting sites suggest that these areas may serve as nesting refugia if management regimes change and/or poaching at continental sites increases. This is underscored by a much greater hatching success at least one insular nesting site (90 percent at the Revillagigedos (Juarez-Ceron et al., 2003) vs. 44.7–79.4 percent in Michoacán (Figueroa et al., 1993). However, we note that hatching success in the Galápagos is 46.0 (Zarate et al., 2013), thus suggesting that factors other than mainland nesting location also play a role in causing low hatching success.

The nesting season in Michoacán runs from October through January (Alvarado-Díaz and Figueroa, 1990); in the Revillagigedos Islands nesting occurs from March through November with a peak in April/May (Brattstrom, 1982; Awbrey et al., 1984) and in the Galapagos, nesting occurs year-round with a peak from January to March (Zarate et al., 2013). Year-round nesting has also been confirmed for some areas in Costa Rica. The presence of year round nesting at these sites, and non-overlapping nesting seasons at others, suggest that the nesting phenology of green turtles in this DPS may help buffer in geologic time against climate change, both in terms

Comment [A10]: But this is insular too. Differences maybe also due to other reasons?
Response: NEW TEXT HAS BEEN ADDED TO THIS EFFECT
of increased mean incubation temperatures on beaches and in terms of impact to storms and other seasonal events.

At the primary nesting beach in Michoacán, Mexico (Colola), the beach slope aspect is extremely steep and the dune surface at which the vast majority of nests are laid is well-elevated. This site is likely buffered against short-term sea level rise as a result from climate change. Many nesting sites are along these protected beach faces, out of tidal surge pathways. Multiple nesting sites in Costa Rica and in the Galapagos Islands are on beaches that are protected from major swell coming in from the ocean.

There is a range of beach shade levels depending on the nesting beach. At some sites such as those in the Revillagigedos Islands and beaches in Mexico, the beaches have little vegetation and nests are commonly laid in full-sun areas. On the other hand, the beaches in Costa Rica are highly shaded and nests are commonly deposited deep in the coastal scrub bushes and trees. There are also intermediate sites, such as those in the Galapagos that have a mix of full sun and shade sites on any given beach. While the exposed beaches are more likely to suffer from the impacts of climate change, those in shaded areas may be subjected to less heating.

15.2.5. Analysis of Factors Listed Under ESA Section 4(a)(1)

Section 4(a)(1) of the ESA lists five factors (threats) that must be considered when determining the status of a species. These factors are:

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.

The following information on these factors/threats pertains to green turtles found within the boundaries of the East Pacific DPS. Because green turtles from this DPS also are found within the boundaries of the Southwest Pacific DPS, the Central West Pacific DPS, and Central North Pacific DPS, the narrative for these regions should also be consulted.

15.2.5.1. Factor A: Destruction or Modification of Habitat or Range

Coastal development, beachfront lighting, and heavy foot traffic consistently affect hatchlings and nesting turtles on a small portion of this DPS. The extent and level of foraging habitat degradation is not known but can affect all life stages of green turtles and is known to occur in a small portion of this DPS.

Terrestrial Zone

Impacts to green turtle habitat are diverse and widespread in the eastern Pacific Ocean and affect both marine and nesting beach habitats. However, their cumulative impacts are less than those occurring in other, more highly populated and industrialized areas outside of the Eastern Pacific.
Although nesting beaches in Costa Rica, Revillagigedos Islands, and the Galapagos Islands are less affected by coastal development than green turtles in other regions around the Pacific, several of the secondary green turtle nesting beaches in México suffer from coastal development. For example, this is especially acute at Maruata, a tourist site with tourist activity and heavy foot traffic during the nesting season (Seminoff, 1994). Nest destruction due to human presence is also a threat on nesting beaches in the Galapagos Islands (Zárate et al., 2006).

**Neritic/Oceanic Zone**

With respect to environmental degradation in the marine environment, coastal habitats along the continental and insular shores of the eastern Pacific are relatively pristine, although green turtles in San Diego Bay, at the north edge of their range, have high levels of contaminants (Komoroske et al., 2011, 2012). Likewise, although difficult to quantify, many coastal habitats are likely modified today due to the depletion of green turtles (Seminoff et al., 2012). Although the impact from ongoing and proposed human activities is difficult to quantify, the recent human population increases in many areas underscores the need to develop and implement management strategies that balance development and economic activities with the needs of green turtles.

**15.2.5.2. Factor B: Overutilization**

Overutilization for commercial purposes likely was a factor that contributed to the historical declines of this DPS. Sea turtles were, and continue to be, harvested primarily for their meat, although other products have served important non-food uses (Mancini and Koch, 2009; J. Seminoff, NMFS, pers. obs., 2012). Sea turtle oil was for many years used as a cold remedy and the meat, eggs and other products have been highly-valued for their aphrodisiacal qualities, beliefs that strongly persist in the countries bordering the East Pacific DPS (Seminoff et al., 2012). A summary of these impacts is given below.

**Egg Harvest**

Decades of egg harvest have impacted many nesting subpopulations in the East Pacific DPS. This harvest has taken many forms, from single families collecting eggs for subsistence use, to 'professional' egg collectors taking every last egg they could get their hands on to sell at market. In some countries and localities, egg harvest has been legal, while in others it is illegal but persistent due to lack of enforcement. Egg harvest is exacerbated by the high monetary value of eggs, a consistent market demand, and severe poverty in many of the countries in the Eastern Pacific Region where sea turtles are found. Egg harvest is a major conservation challenge at several sites in Costa Rica, including Nombre de Jesus and Zapotillal Beaches, where 90 percent of the eggs were taken by egg collectors during one particular study (Blanco, 2010). Egg harvest is also believed to occur at unprotected nesting sites in Mexico, Guatemala, El Salvador, and Nicaragua (NMFS and FWS, 2007).

**Turtle Harvest**

Although it is likely that nesting green turtles are harvested at least on occasion at some nesting beaches, there is no published information relating to this.
Mortality of turtles in foraging habitats continues to be problematic for recovery efforts in the East Pacific DPS. Green turtles are hunted in many areas of northwest Mexico despite legal protection (Nichols et al., 2002; Seminoff et al., 2003; J. Seminoff, NMFS, pers. obs., 2012). Mancini and Koch (2009) described a black market that killed tens of thousands of green turtles each year in the Eastern Pacific Region.

15.2.5.3. Factor C: Disease or Predation

Disease, specifically FP, was not a factor that contributed to the historical decline of this DPS, and the best available data suggest that FP does not pose a current threat to the persistence of this DPS.

FP is virtually non-existent in green turtles within the East Pacific DPS (Koch et al., 2007), although a variant of FP has been found in one green turtle from San Diego Bay, USA (Greenblatt et al., 2005) that shared DNA affinities with the Mexican green turtle stock (P. Dutton, NMFS, unpubl. data). In addition, a few other turtles in San Diego Bay were believed to have the precursor to FP based on eye anomalies (McDonald and Dutton, 1990).

Predation occurs at low levels in the East Pacific DPS. In the Galapagos Islands there is depredation on eggs and hatchlings by feral pigs (Sus sp.) and beetles (order Coleoptera; Zarate et al., 2013). Predation levels are not reported (Zárate et al., 2003; 2006). There are accounts of jaguars (Panthera onca) killing adult female green turtles (L. Fonseca, National University of Costa Rica, unpubl. data, 2009) at beaches in Costa Rica, but this is not a major problem for the DPS.

15.2.5.4. Factor D: Inadequacy of Existing Regulatory Mechanisms

Our review of regulatory mechanisms under Factor D demonstrates that although regulatory mechanisms are in place that should address direct and incidental take of green turtles in the East Pacific DPS, these regulatory mechanisms are insufficient or are not being implemented effectively to address the needs of green turtles. We find that there is a threat from the inadequacy of existing regulatory mechanisms for fishery bycatch and pollution prevention (Factor E), overutilization from legal and illegal takes (Factor B), and impacts to nesting beach and foraging habitat (Factor A).

While conservation efforts for the East Pacific DPS are substantive and improving and may be reflected in the recent increases in the number of nesting females, they still remain inadequate to ensure the long-term viability of the population. For example, while most of the major nesting beaches are monitored, some of the management measures in place are inadequate and may be inappropriate. On some beaches, hatchling releases are coordinated with the tourist industry or nests are being trampled on or are unprotected. The largest threat on the nesting beach, reduced availability of habitat due to heavy armament and subsequent erosion, is just beginning to be addressed but without immediate attention may ultimately result in the demise of the highest density beaches. Further, it is suspected that there are substantial impacts from illegal, unreported, and unregulated fishing, which we are unable to mitigate without additional fisheries
management efforts and international collaborations. While conservation projects for this population have been in place since 1978 for some important areas, efforts in other areas are still being developed to address major threats, including fisheries bycatch and long-term nesting habitat protection.

The management of green turtles is facilitated by a number of regulatory instruments at regional, national, international levels. A summary of the ten international instruments occurring in the East Pacific DPS that relate to green turtle management is provided in section 15.3.2. Hykle (2002) and Tiwari (2002) reviewed the value of some of these international instruments, which vary in their effectiveness. Often, international treaties 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, 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 a 2002 special issue of the Journal of International Wildlife Law and Policy: International Instruments and Marine Turtle Conservation (Hykle, 2002).

Overall, conservation efforts for green turtles in the East Pacific DPS are inconsistent. While there are numerous varied conservation efforts, especially on the primary nesting beaches, many issues remain due to limited enforcement of existing laws and marine protected areas as well as extensive fishery bycatch, especially in coastal waters. The effectiveness and consistency of conservation measures will need to be increased substantially to maximize the recovery potential of this DPS.

15.2.5.5. Factor E: Other Natural or Manmade Factors

The East Pacific DPS of the green turtle is negatively affected by both natural and anthropogenic impacts as described below in Factor E. Within Factor E, we find that fishery bycatch that occurs throughout the eastern Pacific Ocean, particularly bycatch mortality of green turtles from nearshore gill net fisheries, is a significant threat to the persistence of this DPS. Climate change also has the potential to affect this DPS.

Incidental Bycatch in Fishing Gear

Incidental capture in artisanal and commercial fisheries is a significant threat to the survival of green turtles throughout the Eastern Pacific Ocean. The primary gear types involved in these interactions include longlines, drift nets, set nets, and trawl fisheries. These are employed by both artisanal and industrial fleets, and target a wide variety of species including tunas (*Thunnus* sp.), sharks (class Chondrichthyes), sardines (*Sardinella* sp.), swordfish (*Xiphias gladius*), and mahi mahi (*Coryphaena hippurus*).

In the Eastern Pacific Ocean, particularly areas in the southern portion of this DPS, significant bycatch has been reported in artisanal gill net and longline shark and mahi mahi fisheries operating out of Peru (Kelez *et al.*, 2003; Alfaro-Shigueto *et al.*, 2006) and to a lesser extent, Chile (Donoso and Dutton, 2010). The fishing industry in Peru is the second largest economic
activity in the country, and, over the past few years, the longline fishery has rapidly increased. Currently, nearly 600 longline vessels fish in the winter and over 1,300 vessels fish in the summer. During an observer program in 2003/2004, 588 sets were observed during 60 trips, and 154 sea turtles were taken as bycatch. Green turtles were the second most common sea turtle species in these interactions. Of the two fisheries, sea turtle bycatch was highest during the mahi mahi season, with 0.597 turtles/1,000 hooks, while the shark fishery caught 0.356 turtles/1,000 hooks (Alfaro-Shigueto et al., 2008). A separate study by Kelez et al. (2003) reported a significant number of green turtles bycaught. In many cases, green turtles are kept on board for human consumption; therefore, the mortality rate in this artisanal longline fishery is likely high because sea turtles are retained for future consumption or sale.

In northern portions of the DPS range, bycatch in fisheries has been less-well documented. However along the Baja California Peninsula, Mexico, green turtles have been reported as bycatch in several instances. Koch et al., (2006) reported green turtle bycatch-related dead strandings numbering in the hundreds in Bahia Magdalena. In Baja California Sur, Mexico, from 2006–2009 small-scale gill-net fisheries caused massive green turtle mortality at Laguna San Ignacio, where Mancini et al. (2012) estimated that over 1000 turtles were killed each year in nets set for Guitar Fish.

Reduction of bycatch in the East Pacific DPS has been identified as among the highest conservation priorities for sea turtles globally (Wallace et al., 2010a). This impact can be attributed to two general fishing sectors: industrial fleets and artisanal fleets. Bycatch in coastal areas occurs principally in shrimp trawlers, gill nets and bottom longlines (e.g., Orrego and Arauz, 2004). However, since 1996 and 1997 all countries from Mexico to Ecuador declared the use of TEDs as mandatory for all industrial fleets to meet the requirements to export shrimp to the US under the U.S. Magnuson-Stevens Act (Helvey and Fahy, 2012). Since then, bycatch has not been thoroughly evaluated but it is largely known that most fishermen either improperly implement TEDs or remove them entirely from their trawls. As was the case with sea turtle meat and egg collection, an almost total lack of enforcement of bycatch mitigation measures by local authorities only helps to confound the problem.

Additionally, TEDs are not a requirement for artisanal shrimping boats, which with today’s technology are becoming more ‘industrial’ in ability and have been reported to catch large numbers of sea turtles (A. Zavala, Universidad de Sinaloa, pers. comm., 2012). Bottom-set longlines and gill nets, both artisanal and industrial, also interact frequently with sea turtles, and can have devastating mortality rates, such as has been the case in artisanal fisheries of Baja California, Mexico (Peckham et al., 2007). In purse seine fisheries, which typically target tuna and other large pelagic fish species, the highest rate of turtles are captured with “log sets” sea around natural

Pollution

Other threats such as debris ingestion (Seminoff et al., 2002b) and boat strikes (P. Dutton, NMFS, pers. comm., 2012) also affect green turtles in the Eastern Pacific. There are several factors in addition to coastal development and sea turtle hunting that affect green turtles in the Eastern Pacific. Because of the dispersal of green turtles from nesting sites to areas throughout
the East Pacific DPS, human threats found in the region, particularly those that are widespread, have profound impacts on the local breeding population (e.g., global warming, fisheries bycatch, pollution). In addition, red tide poisoning is also a threat to this species (Delgado-Trejo and Alvarado-Figueroa, 2012).

**Climate Change**

Climate change is another factor that has the potential to greatly affect green turtles. Potential impacts of climate change to green turtles include beach erosion from rising sea levels, repeated inundation of nests, skewed hatchling sex ratios from rising incubation temperatures, and abrupt disruption of ocean currents used for natural dispersal during the complex life cycle (Fish et al., 2005; Hawkes et al., 2009; Poloczanska et al., 2009). Although not yet quantified, increasing incubation temperatures may also result in heightened egg and hatchling mortality. Impacts from global climate change induced by human activities are likely to become more apparent in future years (Inter-governmental Panel on Climate Change (IPCC, 2007a).

### 15.3. Summary of Existing Conservation Efforts

There have been important advances in the East Pacific DPS. There are indications that wildlife enforcement branches of local and national governments are stepping up their efforts to enforce existing laws, although successes in stemming sea turtle exploitation through legal channels are few and far between. In addition, there are a multitude of NGOs and conservation networks whose efforts are raising awareness about sea turtle conservation.

The first of these conservation alliances commenced in 1997 when, after years of information exchange about shared populations between the nations of the region, the Central American Regional Network for the conservation of sea turtles was created. The first product that resulted from this collaborative effort was the creation of a national sea turtle network in each country of the region, as well as the development of first hand tools, such as a regional diagnosis, a 10-year strategic plan, a manual of best practices, and four regional training and information workshops for people in the region (e.g., Chacón and Arauz, 2001). This initiative is managed by stakeholders in various sectors (private, non-governmental and governmental) across the region. Like many such initiatives, the Central American Regional Network works under the principle "the benefits and achievements from working in alliance are much higher than those from working alone."

#### 15.3.1. National Legislation and Protection

In addition to the international mechanisms, most of the countries have developed legislation to protect sea turtles and/or nesting habitats. They are summarized below. However, the overall effectiveness of this country specific legislation is unknown at this time.

**Chile**

Perhaps the most important national legislation in Chile for the protection of sea turtles was the presidential decree (No. 225) that was passed in November 1995 that established a closed season
for the harvest of all sea turtles in Chilean waters for 30 years. This Decree was updated in February 2007 to become a permanent closure on the use of sea turtles and their products.

**Colombia**

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

**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 green turtles. In 2013, Costa Rica began the process of banning all shrimp trawling in Costa Rican waters, which may someday pay large dividends towards the protection of green turtles.

**Ecuador**

The most important national legislation in Ecuador for the protection of sea turtles was Law RO 51 passed on 12 December 1990 that protected all sea turtles in Ecuadoran 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 Ecuadoran Shrimp fleet.

**El Salvador**

The first national effort for sea turtle protective legislation occurred on February 4, 2009 with the Official Decree 23:382 which declared a complete and permanent ban on harvesting of turtles and their parts or products therefrom. Prior to that, on August 27, 2004 Official Decree 158:364 established requirements for the use of turtle excluder devices on shrimp vessels operating in El Salvador.

**Guatemala**

The first national effort for sea turtle protection was the Presidential Agreement, October 26th, 1971 that declared the closure of capture, circulation and commercialization of green turtles and their eggs. In 1976 this law was re-extended for protecting green turtles. An additional governmental agreement was passed on February 17th, 1981 that prohibited the capture, circulation, and commercialization of all species of sea turtles that inhabit and reproduce on the Guatemalan coasts.

Other Guatemalan national legislation includes the 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 the CITES species such as sea turtles; 2)
Fisheries Law (Decree 80-2002) that mandates the use of TEDs and establishes greater sanctions for violators of the TED law. This law was updated by the Ministerial Agreement 46-2005.

The most recent update to sea turtle laws in Guatemala was the General Hunting Law Decree 36-2004 that established new regulations affecting the green turtles included controlling activities to curtail poaching and illegal trade of sea turtles and its eggs and the enforcement of TEDs in shrimp boats to reduce the number of accidental deaths (Giron, 2006).

**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**

The most important law for sea turtle protection in Mexico was a 1990 presidential decree, which 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.

An additional law for sea turtle protection was a modification of the official Mexican Regulation NOM-002-PESC-1993 that was passed in 1997 to mandate the responsible management of shrimp fisheries throughout Mexico by implementing the use of turtle excluder devices. In 2004 the Official Mexican Emergency Regulation NOM-EM-007-PESC was passed that provided technical specifications for the turtle excluder devices used by the shrimp trawling fleet in Mexico.

**Nicaragua**

The tradition of consuming turtle eggs is prohibited by law (Law No. 641 and Ministerial Resolution No. 043-2005). However, the harvesting and consumption of turtle eggs continue throughout the coastal areas of the Pacific. However, one vital piece of legislation was the declaration of a protected area for the nesting beaches at which sea turtles lay eggs (including green turtles) 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). There is also Law No. 003 declared on November 18, 2009 which adopts the Code of Conduct for Responsible Organization of the United Nations Food and Agriculture Organization (FAO) and its Annexes Fishing and the International Plan of Action is also taken to Prevent, Deter and Eliminate fishing Illegal Unreported and Unregulated of the United Nations Food and Agriculture Organization (FAO).
In addition, Resolution AG-0095-2009 of February 2009 (G.O. 26230) declares protected areas near Isla Escudo de Veraguas and a portion of Los Mosquitos Gulf in the District of Bocas del Toro, which will be named la “Paisaje Protegido Isla Escudo de Veraguas-Degó.” In Article 9 of this resolution, it is warned that anyone who commits acts against the conservation and sustainable management of natural resources and wildlife of the protected area created by this resolution or violates the environmental regulations, will be sanctioned in accordance to what is established in the current legislation.

**Peru**

Sea turtle protection was first mandated in Peru in 2001 by the Forestry and Wildlife Law 014-2001 that established measures for the protection of wildlife resources and established infractions for wildlife protection violators. This was updated in 2004 by Law 034-2004 that 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 and legislation in the United States that promotes the protection and conservation of sea turtles. The most relevant to sea turtle protection within U.S. Boarders is the U.S. Endangered Species Act of 1973. The U.S. ESA has as its purpose 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. 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 conservation plan (often called HCP for 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. The ESA has been instrumental in curtailing the demise and assisting in the recovery of many species, and green turtles are included among them.

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 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 that sell fisheries products to the US.

The Marine Turtle Conservation Act is also a key element of sea turtle protection in the US and internationally. This Act authorizes a dedicated fund to support marine turtle conservation.
projects in foreign countries, with emphasis on protecting nesting populations and nesting habitat.

15.3.2. International Instruments

Several treaties and/or regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the East Pacific DPS. The international instruments listed below apply to sea turtles found in the Eastern Pacific and are described in Appendix 5.

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 efforts: (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 countries’ 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 that are among the world’s finest for regional fisheries management organizations; (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. A summary of all international instruments that apply to this DPS follows.

- Convention on Biological Diversity
- Convention on the Conservation of Migratory Species of Wild Animals
- Convention on International Trade in Endangered Species of Wild Fauna and Flora
- Convention for the Protection of the Marine Environment and Coastal Area of the South-East Pacific (Lima Convention)
- Convention for the Protection of the Natural Resources and Environment of the South Pacific Region
- Food and Agriculture Organization Technical Consultation on sea turtle-fishery interactions,
- Inter-American Convention for the Protection and Conservation of Sea Turtles
- International Union for the Conservation of Nature
- United States Endangered Species Act
- United States Magnuson-Stevens Conservation and Management Act

15.4. Assessment of Significant Portion of its Range (SPR)

The SRT reviewed the information on threats and extinction risk within each DPS to determine whether there were portions of the range of the DPS that might be considered “significant portions of the range” in accordance with the definition of endangered and threatened species and the draft policy that interprets that term (Section 3.4).
Generally, nesting trends are either stable or increasing throughout the DPS. The only site for which we have robust long-term data is for Michoacán, Mexico, which shows a clear increasing trend. In Central America, though fewer data are available, anecdotal information suggests that there are many more green turtles nesting today than there were a few decades ago. In terms of threats, in-water threats are relatively uniform throughout the range; however, impacts on beaches are not uniform. There is ongoing egg harvest at the nesting sites in Central America, as opposed to the two primary rookeries (Galapagos and Michoacán), where there is virtually 100 percent protection of nesting sites. The SRT concluded that if the Central American rookeries were lost, it would not result in a substantially increased risk of extinction to the DPS as a whole. As such, the SRT concluded that the need to consider a significant portion of the range does not apply to this DPS. See Section 3.4 for more details on the SPR deliberative process.

15.5. Assessment of Extinction Risk

For the SRT's assessment of extinction risk for green turtles in the East Pacific DPS, there were two separate sets of ranking exercises: One focusing on the importance that each SRT member placed on each of the six critical assessment elements for this region (Table 15.3), and a second which reflects the SRT members' expert opinion about the probability that green turtles would fall into any one of six different extinction probability ranges (Table 15.4; see Section 3.3 for discussion of this process).

With respect to the importance rankings for the first four critical assessment elements (Abundance, Trends, Spatial Structure, Diversity/Resilience; Table 15.3), none of the mean scores were greater than 1.7, which means SRT members thought it was unlikely that these element contributes significantly to risk of extinction by themselves, but they did have some concern that these may, in combination with other factors be problematic for green turtles. SRT members had least concern that trends in abundance (mean rank=1.2) contributed to extinction risk, perhaps owing to the fact that the longest data set for green turtle population trends has shown a dramatic increase in nesting numbers over the last 10 yrs. SRT members also generally thought that conservation efforts not yet reflected in the nester abundance by the population weighed slightly heavier in their risk assessment voting than did any threats that may emerge in the future (Table 15.3).
Table 15.3. Summary of ranks that reflect the importance placed by each SRT member on the critical assessment elements considered for the East Pacific DPS. For Elements 1–4, higher ranks indicate higher risk.

<table>
<thead>
<tr>
<th>Critical Assessment Elements</th>
<th>Element 1</th>
<th>Element 2</th>
<th>Element 3</th>
<th>Element 4</th>
<th>Element 5</th>
<th>Element 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>Abundance (1 to 5)</td>
<td>1.60</td>
<td>1.20</td>
<td>1.70</td>
<td>1.70</td>
<td>–0.80</td>
<td>1.00</td>
</tr>
<tr>
<td>Trends / Productivity (1 to 5)</td>
<td>0.27</td>
<td>0.13</td>
<td>0.26</td>
<td>0.30</td>
<td>0.13</td>
<td>0.21</td>
</tr>
<tr>
<td>Spatial Structure (1 to 5)</td>
<td>1–3</td>
<td>1–2</td>
<td>1–3</td>
<td>1–4</td>
<td>(–1)–0</td>
<td>0–2</td>
</tr>
<tr>
<td>Diversity / Resilience (1 to 5)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Five-Factor Analyses (–2 to 0)</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Conservation Efforts (0 to 2)</td>
<td></td>
<td></td>
<td></td>
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</tbody>
</table>

Of the six categories describing the probability that the East Pacific DPS will reach a critical risk threshold within 100 years (Table 15.4), the SRT members voted overwhelmingly for the two lowest probability designations, with 63.6 percent of the votes in the '<1%' range and 16.7 percent of the votes in the '1–5%' range. A total of 0.5 percent of the votes were cast for the highest range (>50%) and 5 percent of the votes were cast in the '21–50%' risk range.

Table 15.4. Summary of Green Turtle SRT member expert opinion about the probability that the DPS will reach a critical risk threshold within 100 years. Each SRT member assigned 100 points across the rank categories. This is a continuum with less risk on the left and more risk on the right.

<table>
<thead>
<tr>
<th>Probability of Reaching Critical Risk Threshold</th>
<th>&lt;1%</th>
<th>1–5%</th>
<th>6–10%</th>
<th>11–20%</th>
<th>21–50%</th>
<th>&gt;50%</th>
</tr>
</thead>
<tbody>
<tr>
<td>MEAN ASSIGNED POINTS</td>
<td>63.64</td>
<td>16.73</td>
<td>6.36</td>
<td>7.82</td>
<td>5.00</td>
<td>0.45</td>
</tr>
<tr>
<td>SEM</td>
<td>11.97</td>
<td>6.97</td>
<td>2.66</td>
<td>5.27</td>
<td>3.50</td>
<td>0.45</td>
</tr>
<tr>
<td>Min</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>Max</td>
<td>97</td>
<td>65</td>
<td>30</td>
<td>50</td>
<td>35</td>
<td>5</td>
</tr>
</tbody>
</table>

In the vote justifications, a relatively high abundance, success of conservation efforts, and positive nesting trends weighed against the uncertainty of spatial structure and diversity, low number of high density beaches, and continual threat of bycatch and climate change.

15.6. Synthesis and Integration

During the analysis of the East Pacific DPS’s status an integrated approach was taken by the SRT to consider the many critical elements described earlier. Nesting abundance was ranked with a low risk of extinction. There were three primary regions considered under the critical assessment element of absolute abundance, with Mexico having the largest number of nesting females (13,664 nesters among seven nesting sites; Table 15.1), followed by Ecuador (3,603
females in the Galapagos, 15 on mainland; Table 15.1), and Costa Rica (2,826 females distributed among 26 nesting sites; Table 15.1).

Although trend information is lacking for the vast majority of sites, based on 25-year trend line for Michoacán, Mexico—the largest nesting aggregation in the East Pacific DPS—it is clear that green turtle nesting has increased since the population's low point in the mid-1980s. This observed increase may have resulted from the onset of nesting beach protection in 1979, as is suggested by the similarity in timing between the onset of beach conservation and the age-to-maturity for green turtles in Pacific Mexico. In addition to Mexico, data from the Galapagos Archipelago does not suggest a declining trend, and the largest-ever nesting numbers reported in Costa Rica suggest this site may be on the increase as well.

The SRT examined four lines of evidence relating to spatial structure of the population, including genetic data, flipper and satellite tagging, and demographic data. The genetic data indicate that there are regional genetic stocks, including Revillagigedos Archipelago (Mexico), Michoacán (Mexico), Central America (Costa Rica) and the Galapagos Islands. To a lesser extent, there was also a level of substructure evident in flipper tag returns, with a clear separation between the northern nesting beaches in Mexico, where most returns occurred north of El Salvador, and the nesting beaches in the Galapagos, where the vast majority of tag returns came from Nicaragua south to Peru. There is a relative paucity of satellite tracking data for green turtles in the East Pacific DPS, both those tracks available for our examination similarly depict separation between northern and southern portions of this DPS. With respect to demographic data, the data available for the nesting sites within this DPS (Michoacán, Revillagigedos Islands, and Galapagos) indicate that these relatively well-studied rookeries differ substantially in key demographic parameters such as mean nesting size, hatching success, and nest size.

The aspects considered under the Diversity / Resilience critical assessment element include the genetic diversity within the DPS, the overall nesting spatial range, the diversity in nesting season, and diversity of nesting site structure and orientation. As mentioned above, there is significant genetic substructuring within this DPS, perhaps suggesting a level of resilience to population genetic bottlenecks. With respect to spatial range of nesting, this DPS has a very broad nesting range, with nesting occurring from the tip of the Baja California Peninsula to Northern Peru. Such a broad latitudinal range may be advantageous to green turtles in this DPS in the face of global climate change. Likewise, with year round nesting at several sites and non-overlapping nesting seasons at other, it appears that this DPS my benefit from nesting season temporal diversity in relation to population resilience. Lastly, with nesting at both continental and insular sites—the latter of which apparently has much lesser human threats—indicates that there are at least some relatively threat-free nesting refugia within this DPS's range.

Green turtles are impacted by a variety of threats in the East Pacific DPS. These include harvest of eggs and turtles for food and non-food uses, bycatch in coastal and offshore marine fisheries gear, coastal development, beachfront lighting, and heavy foot traffic. Although the situation has improved to some extent, the harvest of turtles and their eggs continues throughout much of the range, although more problematic outside of the Galapagos Islands, particularly in Central America (egg harvest) and Mexico (harvest of foraging turtles). Mortality from diseases such as FP is not a problem in the Eastern Pacific, but predation by natural predators is a very large
concern, particularly in the Galapagos and, to a lesser extent, in Costa Rica. Green turtle interactions and mortalities with coastal and offshore fisheries in the eastern Pacific region are of concern and are considered an impediment to green turtle recovery in the East Pacific DPS.

There are 12 countries along the Pacific Coast of the Americas, which marks the eastern boarder of this DPS, and all have some type of sea turtle protection as part of their national legislation. There are also numerous conservation networks in the region and several international instruments that pave the way for sea turtle protection in the East Pacific DPS. NMFS and U.S. NGOs have worked with international entities to assess bycatch mortality, reduce interactions and mortalities in coastal and offshore fisheries, and more than ever are convey information to fishers and other stakeholders through participatory activities, events and outreach. Together, these conservation actions, national laws, and international instruments both inside and outside of the United States have provided the foundation for what appears to be an ongoing population recovery in the region, particularly in Mexico.

Consistent with the nesting numbers and trends summarized above, and in consideration of the other four critical assessment elements, the SRT determined the likelihood of reaching a critical risk threshold within 100 years was relatively low (63.6 percent of votes cast for the ‘<1%’ likelihood category). With the ’1–5%’ category receiving 16.7 percent of the votes, a total of 80.3 percent of votes were for risk categories of 5 percent of less.
16. SYNTHESES

16.1. DPS Consideration

The SRT reviewed the best available scientific information on green turtles, to determine whether DPSs exist, in accordance with the DPS policy.

The policy defines a population to be a DPS if it is both discrete and significant relative to its taxon. A population may be considered discrete if it satisfies either one of the following conditions:

- 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.
- It is delimited by international governmental boundaries within which differences in control of exploitation, management of habitat, conservation status, or regulatory mechanisms exist that are significant in light of section 4(a)(1)(D) of the ESA.

With regard to discreteness, the SRT evaluated genetic evidence, tagging (flipper and PIT tags) and satellite telemetry data, demographics information, oceanographic features, and geographic barriers (Table 16.1).

The SRT then considered whether each of the 11 identified discrete population segments is significant relative to its taxon (Table 16.2).

The following tables are also presented as Tables 4.1 and 4.2.
<table>
<thead>
<tr>
<th>DISCRETENESS</th>
<th>DPS</th>
<th>Spatial Separation</th>
<th>Demography</th>
<th>Tagging</th>
<th>Genetics</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td></td>
<td>physical</td>
<td>physiological</td>
<td>behavioral</td>
<td>genetics</td>
</tr>
<tr>
<td>1. North Atlantic</td>
<td>Some overlap at southern edge of N Atl range (DPS #3), no overlap with Med (DPS #2)</td>
<td>minimal transboundary recoveries (DPS #3, no transboundary tag recoveries DPS #2), localized telemetry (DPS #2,3), distinct FP phylogeny (DPS #3)</td>
<td>N Atl haplotypes found juveniles captured in Brazil and Argentina (DPS #3), no genetic structure from nDNA (DPS #3, but small number of markers)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>2. Mediterranean</td>
<td>Only population in entire sea basin</td>
<td>second smallest MNS of any region (after EP)</td>
<td>no transboundary recoveries (DPS #1), localized telemetry, extensive data, no tracking immigrations (DPS #1)</td>
<td>Clear genetic differences DPS #1</td>
<td></td>
</tr>
<tr>
<td>3. South Atlantic</td>
<td>Some overlap at northern edge of S Atl range (DPS #1)</td>
<td>largest nesters globally</td>
<td>extensive movements within region, but no immigration or emigration revealed with sat telemetry (DPS #1,4), distinct FP phylogeny (DPS #1)</td>
<td>Neutral (DPS #6), Signal, but not strong (DPS #6), no nDNA, just mtDNA (DPS #6), Clear signal (DPS #3,5)</td>
<td></td>
</tr>
<tr>
<td>4. SW Indian</td>
<td>Cape Good Hope (DPS #3), No clear current boundaries (DPS #5,6), Apparent nesting gap (DPS #3,5,6)</td>
<td>nesters larger (DPS #5,6)</td>
<td>minimal transboundary recoveries DPS #5, no transboundary recoveries (DPS #3), no transboundary recoveries and minimal data DPS #6, localized telemetry, extensive data (Bourjea), no immigrations, minimal data (DPS #6)</td>
<td>Globally unique clade, Clear genetic differences (DPS #4,6), almost all rookeries in N Indian un-sampled</td>
<td></td>
</tr>
<tr>
<td>5. N Indian</td>
<td>Apparent nesting gap (DPS #4,6)</td>
<td>nesters smaller (DPS #4,6)</td>
<td>minimal transboundary recoveries (DPS #4), no transboundary recoveries (DPS #6, localized telemetry (Pilcher)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Region</td>
<td>Description</td>
<td>Discreteness</td>
<td>Notes</td>
<td></td>
<td></td>
</tr>
<tr>
<td>-----------</td>
<td>-----------------------------------------------------------------------------</td>
<td>------------------------------------------------------------------------------</td>
<td>----------------------------------------------------------------------</td>
<td></td>
<td></td>
</tr>
<tr>
<td>6. E Indian-W Pacific</td>
<td>Wallace Line - Biogeographic boundary (DPS #7), Apparent nesting gap (DPS #5), Large distance (DPS #4), Oceanographic currents, possible connectivity (DPS #4)</td>
<td>Nesters larger (DPS #5)</td>
<td>moderate transboundary recoveries (DPS #7,8), rare transboundary recoveries (DPS #9), localized telemetry, no transboundary recoveries (DPS #10)</td>
<td>Globally unique haplotypes, Distinct and high nucleotide diversity, Clear genetic differences (DPS #5,7-9), Historical genetic connectivity (DPS #4)</td>
<td></td>
</tr>
<tr>
<td>7. CW Pacific</td>
<td>Wallace Line - Biogeographic boundary (DPS #6), Oceanographic boundary (DPS #10)</td>
<td></td>
<td>Moderate transboundary telemetry, small sample size, Moderate transboundary recoveries (DPS #6), Minimal transboundary recoveries (DPS #8)</td>
<td>Globally unique haplotypes, Clear genetic differences (DPS #6, 8,9), AMOVA supports stand-alone entity. No genetic immigration (DPS #8,9) Dutton pers. comm.</td>
<td></td>
</tr>
<tr>
<td>8. SW Pacific</td>
<td>Closely proximate DPSs</td>
<td></td>
<td>moderate transboundary recoveries (DPS #6), minimal transboundary recoveries (DPS #7,9), localized telemetry, small sample size</td>
<td>Globally unique haplotypes. Oldest Cm haplotype lineages, Distinct and high nucleotide diversity, Clear genetic differences (DPS #6,7,9), nDNA and mtDNA distinctiveness (DPS #6,7,9)</td>
<td></td>
</tr>
<tr>
<td>9. CS Pacific</td>
<td>Oceanographic barrier (DPS #10,11), EP turtles found in Am Samoa longline (DPS #11)</td>
<td>data deficient</td>
<td>modest transboundary recoveries (DPS #7,8), localized telemetry, limited data, minimal transboundary immigrants (DPS #7,8), no recoveries, telemetry movements (DPS #10)</td>
<td>Clear genetic differences (DPS #7,8,10), Only two rookeries sampled</td>
<td></td>
</tr>
<tr>
<td>10. CN Pacific</td>
<td>Most isolated archipelago globally, Oceanographic barrier (DPS #11), Large distances (DPS #7-9,11)</td>
<td>nesters larger (DPS #11)</td>
<td>rare transboundary recoveries, extensive data (DPS #6-9,11), localized telemetry, extensive data</td>
<td>No shared haplotypes (DPS #6-9), but, shared haplotype with EP (DPS #11)</td>
<td></td>
</tr>
</tbody>
</table>
### DISCRETENESS

<p>| 11. Eastern Pacific | Moderate numbers of juveniles found in WP (7,8) and high seas of CNP regions (10), CSP 'yellow' juveniles (9) found in southeastern EP | Smallest mean nesting size of any DPS, They're MOSTLY black | no tag recoveries of EP turtles outside EP, no tag recoveries of WP/CP in EP, no sat tracks of EP turtles outside EP, no sat tracks of EP/WP turtles in EP | Clear genetic differences (7-10), some shared haplotypes (10) |</p>
<table>
<thead>
<tr>
<th>DPS</th>
<th>Ecological Setting</th>
<th>Gap in Range</th>
<th>Marked Genetics</th>
</tr>
</thead>
<tbody>
<tr>
<td>1. North Atlantic</td>
<td>Caribbean sea is highly unique; expansive <em>Thalassia sp.</em> <em>Pinnace</em> seagrass beds, Broad continental shelf</td>
<td>N. Atl is a massively large region, Gene flow with adjacent region (3)</td>
<td>Endpoints are quite distinct based on mtDNA (2,3), Some globally unique haplotypes</td>
</tr>
<tr>
<td>2. Mediterranean</td>
<td>Med is a distinct and unique habitat</td>
<td>Population encompasses large region, Apparent biogeographic boundary (West Med region)</td>
<td>100% globally unique haplotypes, Significant difference in mtDNA markers (1)</td>
</tr>
<tr>
<td>3. South Atlantic</td>
<td>Ascension Isl. is only mid-ocean island nesting site, Pre-Holocene evolutionary links</td>
<td>Population encompasses large region, Apparent biogeographic boundary (equator)</td>
<td>globally unique haplotypes</td>
</tr>
<tr>
<td>4. SW Indian</td>
<td>Mozambique channel, upwelling, Largest nesting size for Indian Ocean **</td>
<td>Tagging and telemetry do not indicated immigration (3,5,6), shared haplotypes could indicate recent connectivity (6), lack of telemetry studies (6), apparent biogeographic barrier (3), no apparent biogeographic barrier (5,6)</td>
<td>Largest nesting size for Indian Ocean **, Significant difference in mtDNA markers (3,5), NS difference in mtDNA markers (6)</td>
</tr>
<tr>
<td>5. N Indian</td>
<td>Distinct and Unique habitat, (heat adapted coral in Persian and Red Seas)</td>
<td></td>
<td></td>
</tr>
<tr>
<td>6. E Indian- W Pacific</td>
<td>Most extensive continental shelf globally</td>
<td>Population encompasses large region, Loss would create major connectivity gap (4-5 to 7-8)</td>
<td>Ancestral haplotypes, significant mtDNA diversity</td>
</tr>
<tr>
<td>7. CW Pacific</td>
<td></td>
<td>Population encompasses large region, Apparent oceanic boundary (6,10), Apparent biogeography boundary (6)</td>
<td>Globally unique haplotypes</td>
</tr>
</tbody>
</table>

**Comment [A2]:** Comment: N IndianEcological setting: Spelling mistake “corral”. These coral habitats (not extensively used by green turtles) occur only less than half the marine area of the N Indian DPS, with most of the nesting taking place outside these areas. Though they are important habitats, the lack of association with green turtles puts into question their inclusion as an ecological setting for green turtles in this DPS.

5. N Indian Genetics: The Saudi haplotypes may not have been found elsewhere but does this justify indicating the whole DPS (the rest of which has not been sampled) should be considered distinct, or perhaps only the Saudi/Gulf turtles? The main text is correctly far more cautious about classifying this DPS on genetic grounds.

Response: The table is summary support info, so we do not suggest that genetics alone justifies designation of whole DPS, but it provides strong evidence for a regional distinction to consider along with other elements that are discussed with caveats in the text body of the report. I have revised the text without altering the tone of the information that was presented for voting consideration.
### SIGNIFICANCE

<table>
<thead>
<tr>
<th>Region</th>
<th>Significance</th>
<th>Notes</th>
</tr>
</thead>
<tbody>
<tr>
<td>8. SW Pacific</td>
<td>GBR, periodic isolation over geological time</td>
<td>Periodic isolation over geological time.</td>
</tr>
<tr>
<td></td>
<td>Central region adjacent to 6, 7, 9, but apparent ability to emigrate from 6,7,9</td>
<td>Ancient lineage, significant mtDNA diversity</td>
</tr>
<tr>
<td>9. CS Pacific</td>
<td>Population encompasses large oceanic region, Other proximate DPSs</td>
<td>A single, globally unique haplotype.</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Extensive sampling in other regions has not detected CSPac haplotype</td>
</tr>
<tr>
<td>10. CN Pacific</td>
<td>no continental shelf, mid basin oceanic pinnacles, basking</td>
<td>Population encompasses large oceanic region, significant bridge for colonizing EP(11)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Globally unique haplotypes, Extensive sampling in other regions has not detected haplotypes</td>
</tr>
<tr>
<td>11. Eastern Pacific</td>
<td>unique diet, very narrow continental shelf, very low levels of seagrass, equatorial upwelling (ENSO)</td>
<td>Huge range, apparent historic gene flow (10)</td>
</tr>
<tr>
<td></td>
<td></td>
<td>Globally unique haplotypes, Extensive sampling in other regions has not detected haplotypes, smallest MNS of all regions</td>
</tr>
</tbody>
</table>

#### 16.2. Critical Assessment Elements

After the 11 DPSs were identified, the SRT assessed the extinction risk for each DPS. Six critical assessment elements were considered and quantified in this assessment: (1) abundance; (2) population growth rate or productivity; (3) spatial structure; (4) diversity / resilience; (5) threats (as represented by the five factors in section 4(a)(1) of the ESA); and (6) conservation efforts.

Using the the guidelines presented in McElhany et al. (2000) when considering the contribution of each of the population elements to the risk of extinction of a given DPS. These guidelines include an analytical look at abundance, trends, and spatial structure.

With regard to abundance (Table 16.3), the SRT used the following guidelines:

- A population should be large enough to have a high probability of surviving environmental variation of the patterns and magnitudes observed in the past and expected in the future;
- a population should have sufficient abundance for compensatory processes to provide resilience to environmental and anthropogenic perturbation;
- a population should be sufficiently large to maintain its genetic diversity over the long term;
- a population should be sufficiently abundant to provide important ecological functions throughout its life-cycle; and
• population status evaluations should take uncertainty regarding abundance into account.

Table 16.3. Summary of green turtle nester abundance distribution for each DPS.

<table>
<thead>
<tr>
<th>TOTAL NESTER ABUNDANCE</th>
<th>N Atlantic</th>
<th>Mediterranean</th>
<th>S Atlantic</th>
<th>SW Indian</th>
<th>N Indian</th>
<th>E En/W Pacific</th>
<th>C W Pacific</th>
<th>SW Pacific</th>
<th>C S Pacific</th>
<th>C N Pacific</th>
<th>E Pacific</th>
<th>TOTALS</th>
</tr>
</thead>
<tbody>
<tr>
<td>unquantified</td>
<td>26</td>
<td>0</td>
<td>37</td>
<td>23</td>
<td>1</td>
<td>7</td>
<td>16</td>
<td>1</td>
<td>21</td>
<td>0</td>
<td>4</td>
<td>136</td>
</tr>
<tr>
<td>1-10</td>
<td>17</td>
<td>21</td>
<td>0</td>
<td>2</td>
<td>5</td>
<td>7</td>
<td>6</td>
<td>0</td>
<td>11</td>
<td>5</td>
<td>8</td>
<td>82</td>
</tr>
<tr>
<td>11-50</td>
<td>6</td>
<td>5</td>
<td>0</td>
<td>0</td>
<td>5</td>
<td>8</td>
<td>9</td>
<td>0</td>
<td>11</td>
<td>6</td>
<td>11</td>
<td>61</td>
</tr>
<tr>
<td>51-100</td>
<td>3</td>
<td>3</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>4</td>
<td>6</td>
<td>0</td>
<td>7</td>
<td>0</td>
<td>6</td>
<td>31</td>
</tr>
<tr>
<td>101-500</td>
<td>10</td>
<td>5</td>
<td>3</td>
<td>7</td>
<td>4</td>
<td>8</td>
<td>0</td>
<td>2</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>28</td>
</tr>
<tr>
<td>501-1000</td>
<td>6</td>
<td>0</td>
<td>3</td>
<td>1</td>
<td>5</td>
<td>7</td>
<td>2</td>
<td>4</td>
<td>1</td>
<td>1</td>
<td>2</td>
<td>32</td>
</tr>
<tr>
<td>1001-5000</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>5</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>8</td>
</tr>
<tr>
<td>5001-10000</td>
<td>1</td>
<td>0</td>
<td>2</td>
<td>4</td>
<td>2</td>
<td>1</td>
<td>0</td>
<td>3</td>
<td>0</td>
<td>0</td>
<td>1</td>
<td>14</td>
</tr>
<tr>
<td>&gt;100000</td>
<td>1</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>0</td>
<td>1</td>
</tr>
<tr>
<td>TOTAL # SITES in DPS</td>
<td>74</td>
<td>32</td>
<td>51</td>
<td>17</td>
<td>38</td>
<td>58</td>
<td>51</td>
<td>12</td>
<td>57</td>
<td>12</td>
<td>36</td>
<td>438</td>
</tr>
<tr>
<td>TOTAL NESTER ABUNDANCE</td>
<td>167,528</td>
<td>404,992</td>
<td>63,335</td>
<td>84,199</td>
<td>55,243</td>
<td>77,099</td>
<td>6,518</td>
<td>83,058</td>
<td>2,902</td>
<td>3,846</td>
<td>20,112</td>
<td>564,739</td>
</tr>
</tbody>
</table>

LARGEST ROOKERY

<table>
<thead>
<tr>
<th></th>
<th>Tortuguero, Costa Rica</th>
<th>Akyatan, Turkey</th>
<th>Polillo, Guinea Bissau</th>
<th>Europa Island</th>
<th>Ras Shammar, Yemen</th>
<th>Wellesley Group, Australia</th>
<th>Ulithi Atoll, FSM</th>
<th>Capricorn Group, Micronesia</th>
<th>Scilly Atoll, Fr.</th>
<th>Polynesia</th>
<th>French Frigate Shoals</th>
<th>Colombia, Mexico</th>
</tr>
</thead>
<tbody>
<tr>
<td>% at LARGEST ROOKERY</td>
<td>79%</td>
<td>25%</td>
<td>40%</td>
<td>30%</td>
<td>33%</td>
<td>32%</td>
<td>22%</td>
<td>37%</td>
<td>36%</td>
<td>96%</td>
<td>58%</td>
<td></td>
</tr>
</tbody>
</table>

Comment [A3]: Comment: Nester abundance scale seems to be missing a level (10001-50000) and instead is lumped into 10001-10000 which is effectively a grouping of 1 whole order of magnitude and this seems excessive. Suggest revising this grouping system and re-presenting data throughout the text in all sections.

Response: Table has been revised.

N Indian shows total 56,517 whereas the total presented in Table 9.1 is 56,917. Ras Sharma contributes 31.6% according to value in Table 9.1 but is shown as 31.8% in Table 16.3. Data need checking for accuracy.

Response: Table has been revised.

Comment [A4]: Comment: if you update the Cook Islands nesting abundance and it ends up in the next group 501-1000. Will you please adjust the Table 16.3? It probably won’t change the overall abundance estimate of 3600-3700 females, so we could probably leave it this time round. By the next review I should have tag and demographic data. I’m already doing some post-hatching nest inventories, but early days still.

Response: Table has been revised.
With regard to population growth or productivity (Table 16.4) the SRT used the following guidelines:

- A population’s natural productivity should be sufficient to maintain its abundance above the viable levels, even during poor ocean conditions;
- a viable population should not exhibit trends or shifts in traits that portend declines in population growth rate; and
- population status evaluations should take into account uncertainty in estimates of population growth rate and productivity-related elements.

Table 16.4. Nesting sites in each DPS with 10 years or more of recent data available, used to determine nesting population trends. Nesting population trend symbols: ▲ = increasing; ▼ = decreasing; — = stable; ? = unknown. Only those nesting sites used in bar plots and PVAs are listed below.

<table>
<thead>
<tr>
<th>DPS</th>
<th>Nesting Site</th>
<th>Bar plot / PVA</th>
<th>Population Trend</th>
</tr>
</thead>
<tbody>
<tr>
<td>DPS #1: North Atlantic</td>
<td>El Cuyo, Mexico</td>
<td>Bar plot</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>San Felipe, Cuba</td>
<td>Bar plot</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>Guanal, Cuba</td>
<td>Bar plot</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>Tortuguero, Costa Rica</td>
<td>PVA</td>
<td>▲</td>
</tr>
<tr>
<td></td>
<td>Isla Aquada, Mexico</td>
<td>PVA</td>
<td>▲</td>
</tr>
<tr>
<td></td>
<td>Guanahacabibes, Cuba</td>
<td>PVA</td>
<td>▲</td>
</tr>
<tr>
<td></td>
<td>Index Beach, Florida</td>
<td>PVA</td>
<td>▲</td>
</tr>
<tr>
<td>DPS #2: Mediterranean</td>
<td>Akrotiri, Cyprus</td>
<td>Bar plot</td>
<td>?</td>
</tr>
<tr>
<td></td>
<td>North Karpaz, Cyprus</td>
<td>Bar plot</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>Akyatan, Turkey</td>
<td>Bar plot</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>Kazanli, Turkey</td>
<td>Bar plot</td>
<td>▼</td>
</tr>
<tr>
<td></td>
<td>Israel</td>
<td>Bar plot</td>
<td>▲</td>
</tr>
<tr>
<td></td>
<td>Samandag, Turkey</td>
<td>Bar plot</td>
<td>?</td>
</tr>
<tr>
<td></td>
<td>West Coast, Cyprus</td>
<td>PVA</td>
<td>▼</td>
</tr>
<tr>
<td>DPS #3: South Atlantic</td>
<td>Ascension Island, UK</td>
<td>Bar plot</td>
<td>?</td>
</tr>
<tr>
<td></td>
<td>Galibi Reserve and Matapica, Suriname</td>
<td>Bar plot</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>Atol das Rocas, Brazil</td>
<td>Bar plot</td>
<td>—</td>
</tr>
<tr>
<td>DPS #4: Southwest Indian</td>
<td>Glorieuses, Eparses Islands, France</td>
<td>Bar plot</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>Europa, Eparses Island, France</td>
<td>Bar plot</td>
<td>—</td>
</tr>
<tr>
<td></td>
<td>Tromelin, Eparses Islands, France</td>
<td>Bar plot</td>
<td>—</td>
</tr>
</tbody>
</table>

Comment [A5]: Comment: I had hoped to look at the actual PVA results (probabilities of reaching either threshold) in table 16.4. The up and down arrows don’t give a sense of what these risk estimates actually are, and I don’t think they were anywhere else either. I also wondered if the up and down arrows are significant upward or downward trends, or something more subjective. I would suggest making these results more explicit. I had hoped to compare these (loosely) with the results of the voting on extinction risk, but couldn’t really do this to see how other factors outweighed or reinforced what these simulation results showed.

Response: Tables clarified which trend reflected PVA analysis and which were the bar plots using the most recent nesting data.
<table>
<thead>
<tr>
<th>DPS</th>
<th>Nesting Site</th>
<th>Bar plot / PVA</th>
<th>Population Trend</th>
</tr>
</thead>
<tbody>
<tr>
<td>DPS #5: North Indian</td>
<td>Duran Beach, Jiwani, Pakistan</td>
<td>Bar plot</td>
<td>▼</td>
</tr>
<tr>
<td></td>
<td>Zabargard, Egypt</td>
<td>Bar plot</td>
<td>▼</td>
</tr>
<tr>
<td>DPS #6: East Indian-West Pacific</td>
<td>Wan-an, Taiwan</td>
<td>Bar plot</td>
<td>▼</td>
</tr>
<tr>
<td></td>
<td>Lanya, Taiwan</td>
<td>Bar plot</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Sabah Turtle Islands, Malaysia</td>
<td>PVA</td>
<td>▲</td>
</tr>
<tr>
<td></td>
<td>Royal Navy Center, Thailand</td>
<td>PVA</td>
<td>▼</td>
</tr>
<tr>
<td></td>
<td>Redang, Terrenggana, Malaysia</td>
<td>PVA</td>
<td>▼</td>
</tr>
<tr>
<td></td>
<td>Thameehla Island, Myanmar</td>
<td>PVA</td>
<td>▼</td>
</tr>
<tr>
<td>DPS #7: Central West Pacific</td>
<td>Chichijima, Japan</td>
<td>PVA</td>
<td>▲</td>
</tr>
<tr>
<td>DPS #8: Southwest Pacific</td>
<td>Raine Island, Australia</td>
<td>PVA</td>
<td>▲</td>
</tr>
<tr>
<td></td>
<td>Heron Island, Australia</td>
<td>PVA</td>
<td>▲</td>
</tr>
<tr>
<td>DPS #9: Central South Pacific</td>
<td>not available</td>
<td></td>
<td>?</td>
</tr>
<tr>
<td>DPS #10: Central North Pacific</td>
<td>East Island, Hawaii</td>
<td>PVA</td>
<td>▲</td>
</tr>
<tr>
<td>DPS #11: East Pacific</td>
<td>Colola, Mochoacan, Mexico</td>
<td>PVA</td>
<td>▲</td>
</tr>
</tbody>
</table>

With regard to population spatial structure (Table 16.5), the SRT used the following guidelines:

- Habitat patches should not be destroyed faster than they are naturally created;
- some habitat patches should be maintained that appear to be suitable or marginally suitable, but currently contain no individuals;
- source subpopulations should be maintained; and
- analyses of population spatial processes should take uncertainty into account.
Table 16.5. Genetic, tagging, and demographic data for each DPS, used to determine level of spatial structure.

<table>
<thead>
<tr>
<th>DPS #1: North Atlantic</th>
<th>Genetic data</th>
<th>Flipper/satellite tagging</th>
<th>Demographic data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Shallow population structuring</td>
<td>Low population structuring</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>DPS #2: Mediterranean</th>
<th>Genetic data</th>
<th>Flipper/satellite tagging</th>
<th>Demographic data</th>
</tr>
</thead>
<tbody>
<tr>
<td>No population structuring</td>
<td>Similar migration pattern</td>
<td>Consistent parameters, small nesting turtles</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>DPS #3: South Atlantic</th>
<th>Genetic data</th>
<th>Flipper/satellite tagging</th>
<th>Demographic data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Shared haplotype, strong reproductive isolation from other nesting sites, shared haplotype with foraging N. Atlantic</td>
<td>Transoceanic developmental migrations, the wide range of the DPS and the interconnectedness of the different regions</td>
<td>Vary widely among nesting sites, substantial spatial structuring</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>DPS #4: Southwest Indian</th>
<th>Genetic data</th>
<th>Flipper/satellite tagging</th>
<th>Demographic data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Moderate spatial structuring</td>
<td>Green turtles nesting along the East African coast confine their migration to along the coast.</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>DPS #5: North Indian</th>
<th>Genetic data</th>
<th>Flipper/satellite tagging</th>
<th>Demographic data</th>
</tr>
</thead>
<tbody>
<tr>
<td>One stock (Saudi Arabia) has been characterized based on limited sampling and it was found to be very distinct from other nesting sites elsewhere in Indian Ocean</td>
<td>Foraging within Indian ocean</td>
<td>Varies, substantial spatial structuring</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>DPS #6: East Indian-West Pacific</th>
<th>Genetic data</th>
<th>Flipper/satellite tagging</th>
<th>Demographic data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Complex population structure, few common and widespread haplotypes</td>
<td>Broad migration distribution and numerous potential foraging areas</td>
<td>Vary widely among nesting sites, substantial spatial structuring</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>DPS #7: Central West Pacific</th>
<th>Genetic data</th>
<th>Flipper/satellite tagging</th>
<th>Demographic data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Nesting sites separated by more than 1,000 km were significantly differentiated from each other</td>
<td>Nesting females migrate to areas within and outside of the Central West Pacific</td>
<td>Variation suggests substantial spatial structuring</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>DPS #8: Southwest Pacific</th>
<th>Genetic data</th>
<th>Flipper/satellite tagging</th>
<th>Demographic data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Substantial spatial structuring</td>
<td>Foraging is widely dispersed throughout this DPS and also into other DPSs</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th>DPS #9: Central South Pacific</th>
<th>Genetic data</th>
<th>Flipper/satellite tagging</th>
<th>Demographic data</th>
</tr>
</thead>
<tbody>
<tr>
<td>Substantial spatial structuring</td>
<td>Post-nesting females travel the complete geographic breadth of this DPS</td>
<td>No structuring of traits within the DPS</td>
<td></td>
</tr>
</tbody>
</table>
### Spatial Structure

<table>
<thead>
<tr>
<th>DPS #10: Central North Pacific</th>
<th>Low level of spatial structuring</th>
<th>Post-nesting females in the NWHI return to their foraging grounds in the MHL, and that foraging remains exclusively within geographic boundaries of this DPS</th>
<th>No structuring of traits within the DPS</th>
</tr>
</thead>
<tbody>
<tr>
<td>DPS #11: East Pacific</td>
<td>Substantial spatial structuring</td>
<td>Track clustering in Northwest Mexico to Southern United States, and in the Southeast Pacific, from the Galapagos Islands to the high seas</td>
<td>Regional variation</td>
</tr>
</tbody>
</table>

With regard to diversity and resilience (Table 16.6), the SRT used the following guidelines:

- Human-caused factors should not substantially alter variation in traits such as age structure, size, fecundity, morphology, behavior, and molecular genetic characteristics;
- Natural processes of dispersal should be maintained;
- Human-caused factors should not substantially alter the rate of gene flow among populations;
- Natural processes that cause ecological variation should be maintained; and
- Population status evaluations should take uncertainty about requisite levels of diversity into account.
Table 16.6. Spatial range, nesting season, nest site, and genetic diversity for each DPS, used to determine level of diversity and resilience.

<table>
<thead>
<tr>
<th>DPS</th>
<th>Spatial range</th>
<th>Nesting season</th>
<th>Nest Site</th>
<th>Genetic Diversity</th>
</tr>
</thead>
<tbody>
<tr>
<td>#1: North Atlantic</td>
<td>Widespread</td>
<td>Similar</td>
<td>continental and island</td>
<td>Shallow regional substructuring</td>
</tr>
<tr>
<td>#2: Mediterranean</td>
<td>Limited</td>
<td>Similar</td>
<td>insular and continental</td>
<td>Low population substructuring</td>
</tr>
<tr>
<td>#3: South Atlantic</td>
<td>Widespread</td>
<td>Similar</td>
<td>continental and island</td>
<td>Shallow structuring and are all dominated by a common shared haplotype</td>
</tr>
<tr>
<td>#4: Southwest Indian</td>
<td>Limited</td>
<td>Year-round with peak in June that vary</td>
<td>Mostly islands, atolls, and mainland of Africa</td>
<td>High diversity and a mix of unique and rare haplotypes, as well as common and widespread haplotypes</td>
</tr>
<tr>
<td>#5: North Indian</td>
<td>Moderately dispersed</td>
<td>Varies</td>
<td>Continental and island</td>
<td>Limited sampling of single rookery very distinct from other rookeries elsewhere in the Indian Ocean</td>
</tr>
<tr>
<td>#6: East Indian-West Pacific</td>
<td>Widespread</td>
<td>Varies</td>
<td>Continental and island</td>
<td>Varying levels of spatial structure characterized by the presence of rare/unique haplotypes at most rookeries</td>
</tr>
<tr>
<td>#7: Central West Pacific</td>
<td>Moderately dispersed</td>
<td>Varies</td>
<td>Various islands and atolls</td>
<td></td>
</tr>
<tr>
<td>#8: Southwest Pacific</td>
<td>Widely dispersed throughout the region</td>
<td>Year-round with peak</td>
<td>Coral and rocky reefs, sea grass meadows and algal turfs on sand and mud flats</td>
<td>High genetic diversity</td>
</tr>
<tr>
<td>#9: Central South Pacific</td>
<td>Widely dispersed</td>
<td>Varies</td>
<td>Low-lying coral atolls or oceanic islands</td>
<td>Moderate level of diversity and presence of unique haplotypes</td>
</tr>
<tr>
<td>#10: Central North Pacific</td>
<td>Limited</td>
<td>Similar</td>
<td>Low-lying coral atoll</td>
<td>Low level of stock substructuring</td>
</tr>
<tr>
<td>#11: East Pacific</td>
<td>Limited</td>
<td>Differ within DPS</td>
<td>Substantial nesting at both insular and continental nesting</td>
<td>Presence of rare/unique haplotypes</td>
</tr>
</tbody>
</table>

Comment [A6]: Comment: DPS#5: Again the assertion of the whole DPS being "very distinct from other rookeries elsewhere in the Indian Ocean" based on limited sampling from a single location in a geographically marginal rookery is likely to be an overstatement. Moderate the emphasis of this distinctness and suggest 'potential for' or 'indications of' distinctiveness of the whole DPS.

Comment [A7]: Response: The reviewer has misinterpreted the intent of this table...it was NOT to demonstrate discreetness (that was done previously), but to provide a measure of diversity/resilience. I think in our discussion we pointed out that sampling was limited, with only 1 rookery sampled, so it was not possible to determine level of sub-structuring...I've modified text in table to reflect this.

Comment [A8]: Comment: Not true! See my comments on seasonality in section #8.
Response: Nesting occurs throughout the year with peaks that vary between rookeries (Dalleau et al. 2012; Mortimer 2012).
The SRT then assessed threat levels for each DPS. Threat levels were estimated by life stages and habitats, and were grouped into the five factors in section 4(a)(1) of the ESA (Table 16.7):

(A) the present or threatened destruction, modification, or curtailment of its habitat or range;
(B) overutilization for commercial, recreational, scientific, or educational purposes;
(C) disease or predation;
(D) the inadequacy of existing regulatory mechanisms; or
(E) other natural or manmade factors affecting its continued existence.
Table 16.7. Known factors / threats, extent, life stage affected, and level of the threat, presented by DPS and ESA Factor.

<table>
<thead>
<tr>
<th>North Atlantic</th>
<th>KNOWN THREATS</th>
<th>EXTENT</th>
<th>LIFE STAGE AFFECTED</th>
<th>LEVEL</th>
</tr>
</thead>
<tbody>
<tr>
<td>FACTOR A: Habitat</td>
<td>Coastal Development with lighting (including armoring, jetties)</td>
<td>2</td>
<td>1, 2, 3</td>
<td>3, 6</td>
</tr>
<tr>
<td></td>
<td>Erosion from storm events and sand mining</td>
<td>1</td>
<td>1, 2</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Beach engineering</td>
<td>2</td>
<td>1, 2</td>
<td>2, 6</td>
</tr>
<tr>
<td></td>
<td>Climate change: Sea level rise and increased storm events- loss of habitat</td>
<td>1</td>
<td>1, 2</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Beach Driving</td>
<td>3</td>
<td>1, 2</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Fishing practices and anchor damage</td>
<td></td>
<td>4, 5</td>
<td>5</td>
</tr>
<tr>
<td>FACTOR B: Overutilization</td>
<td>Historic-intentional harvest</td>
<td>1</td>
<td>1, 2, 4, 5</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>Current Intentional harvest</td>
<td>3</td>
<td>1, 2, 4, 5</td>
<td>1, (possibly 4 for Nicaragua)</td>
</tr>
<tr>
<td>FACTOR C: Disease</td>
<td>FP</td>
<td>1</td>
<td>4, 5</td>
<td>3 (new FP in TX)</td>
</tr>
<tr>
<td>FACTOR C: Predation</td>
<td>Beach and water</td>
<td>1</td>
<td>1, 2, 3, 4</td>
<td>6</td>
</tr>
</tbody>
</table>

Comment [A9]: Comment: In all Section 16 tables: What makes the difference for LEVEL between “1. Present” and “5. Unknown if Increasing or Decreasing”, as “Present” would indicate it is occurring with no knowledge of trends. Suggest changing classification text.

AML: The numbers do not reflect degrees. It is only an indication of areas where the literature summarizes whether the threat is any of these categories.
# North Atlantic

<table>
<thead>
<tr>
<th>FACTOR D: Inadequacy of Regulation</th>
<th>KNOWN THREATS</th>
<th>EXTENT</th>
<th>LIFE STAGE AFFECTED</th>
<th>LEVEL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Country</td>
<td>MARPOL-implementation and enforcement</td>
<td>3, 4, 5</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Country</td>
<td>Cayman- size limit-between 40 and 6. Haiti- regulations ignored, Nicaragua- consuming turtle eggs prohibited but continues. Panama- egg use and harvest</td>
<td>4</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Local</td>
<td>FL- CCL control line- armoring continues. Lighting and beach furniture- depend on funding for compliance and commitment by County or Municipality</td>
<td>1, 2, 3</td>
<td>1</td>
<td></td>
</tr>
</tbody>
</table>

## Factor E: Other

<table>
<thead>
<tr>
<th>Incidental Bycatch in Fishing Gear</th>
<th>1</th>
<th>4, 5</th>
<th>5,6 (TED)</th>
</tr>
</thead>
<tbody>
<tr>
<td>Vessel Strikes</td>
<td>2</td>
<td>4, 5</td>
<td>5</td>
</tr>
<tr>
<td>Climate Change</td>
<td>1</td>
<td>1, 2, 3, 4, 5</td>
<td>3</td>
</tr>
<tr>
<td>Natural Disasters</td>
<td>1</td>
<td>1, 2, 4</td>
<td>3</td>
</tr>
<tr>
<td>Contaminants</td>
<td>1</td>
<td>3, 4, 5</td>
<td>3</td>
</tr>
<tr>
<td>Dredging</td>
<td>2</td>
<td>4, 5</td>
<td>2</td>
</tr>
</tbody>
</table>

Comment [A9]: Comment: In all Section 16 tables: What makes the difference for LEVEL between “1. Present” and “5. Unknown if Increasing or Decreasing”, as “Present” would indicate it is occurring with no knowledge of trends. Suggest changing classification text.

AML: The numbers do not reflect degrees. It is only an indication of areas where the literature summarizes whether the threat is any of these categories.

Comment [A10]: Panama? Egg use and harvest?

Response: revised
<table>
<thead>
<tr>
<th>Mediterranean</th>
<th>KNOWN THREATS</th>
<th>EXTENT</th>
<th>LIFE STAGE</th>
<th>LEVEL</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Coastal Development with lighting</td>
<td>1</td>
<td>1, 2, 3</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Erosion from storm events and sand extraction and existing jetty</td>
<td>2</td>
<td>1, 2</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Marine Pollution</td>
<td>1</td>
<td>1, 4, 5</td>
<td>2</td>
</tr>
<tr>
<td>FACTOR A: Habitat</td>
<td>Climate change: Sea level rise and increased storm events-loss of habitat</td>
<td>1</td>
<td>1, 2, 3, 4, 5</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Beach Driving</td>
<td>2</td>
<td>1, 2, 3</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Human activity</td>
<td>2</td>
<td>1, 2, 3</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Trawling</td>
<td>1</td>
<td>4, 5</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Fishing practices</td>
<td>2</td>
<td>4, 5</td>
<td>2</td>
</tr>
<tr>
<td>FACTOR B: Overutilization</td>
<td>Historic-intentional harvest</td>
<td>1</td>
<td>1, 2, 4, 5</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>Current Intentional harvest</td>
<td>3</td>
<td>1, 2, 4, 5</td>
<td>1, (Egypt)</td>
</tr>
<tr>
<td>FACTOR C: Disease</td>
<td>FP</td>
<td>4</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>FACTOR C: Predation</td>
<td>Beach</td>
<td>1</td>
<td>1, 2, 3</td>
<td>6</td>
</tr>
<tr>
<td>Mediterranean</td>
<td>KNOWN THREATS</td>
<td>EXTENT</td>
<td>LIFE STAGE</td>
<td>LEVEL</td>
</tr>
<tr>
<td>---------------</td>
<td>---------------</td>
<td>--------</td>
<td>------------</td>
<td>-------</td>
</tr>
</tbody>
</table>

**FACTOR D: Inadequacy of Regulation**

<table>
<thead>
<tr>
<th></th>
<th>Current international- all countries participate, some, none</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Country</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Local</td>
<td></td>
</tr>
</tbody>
</table>

**FACTOR E: Other**

<table>
<thead>
<tr>
<th></th>
<th>Incidental Bycatch in Fishing Gear</th>
<th>1</th>
<th>4, 5</th>
<th>2</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Vessel Strikes</td>
<td>2</td>
<td>4</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Power Generation Activity</td>
<td>2</td>
<td>4, 5</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Pollution</td>
<td>1</td>
<td>3, 4</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Climate Change</td>
<td>1</td>
<td>2, 3, 4, 5</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Natural Disasters</td>
<td>1</td>
<td>1, 2, 3</td>
<td>3</td>
</tr>
<tr>
<td>South Atlantic</td>
<td>KNOWN THREATS</td>
<td>EXTENT</td>
<td>LIFE STAGE</td>
<td>LEVEL</td>
</tr>
<tr>
<td>----------------</td>
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<td>-------</td>
</tr>
<tr>
<td>FACTOR A: Habitat</td>
<td>Coastal Development with lighting including armoring and jetties</td>
<td>2</td>
<td>1, 2, 3</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Erosion from storm events and sand extraction</td>
<td>1</td>
<td>1, 2</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Beach engineering</td>
<td>2</td>
<td>1, 2</td>
<td>2, 6</td>
</tr>
<tr>
<td></td>
<td>Climate change: Sea level rise and increased storm events- loss of habitat</td>
<td>1</td>
<td>1, 2</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Beach and pedestrian traffic</td>
<td>3</td>
<td>1, 2</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>Beach and marine pollution (runoff and sedimentation)</td>
<td>1</td>
<td>2, 3, 4, 5</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td>Non-Native vegetation</td>
<td>2</td>
<td>1, 2</td>
<td>1</td>
</tr>
<tr>
<td></td>
<td>Fishing practices</td>
<td>2</td>
<td>4, 5</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td>Dredging</td>
<td>2</td>
<td>4, 5</td>
<td>2</td>
</tr>
<tr>
<td>FACTOR B: Overutilization</td>
<td>Historic-intentional harvest</td>
<td>1</td>
<td>1, 2, 4, 5</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>Current Intentional harvest</td>
<td>2</td>
<td>1, 2, 4, 5</td>
<td>1, (Northeast Brazil, Suriname, Bioko, Guinea-Bissau, Orange National Park)</td>
</tr>
<tr>
<td>FACTOR C: Disease</td>
<td>FP</td>
<td>1</td>
<td>4, 5</td>
<td>3</td>
</tr>
<tr>
<td>FACTOR C: Predation</td>
<td>Beach and water</td>
<td>1</td>
<td>1, 2, 3, 4, 5</td>
<td>6</td>
</tr>
</tbody>
</table>
### South Atlantic

<table>
<thead>
<tr>
<th>KNOWN THREATS</th>
<th>EXTENT</th>
<th>LIFE STAGE</th>
<th>LEVEL</th>
</tr>
</thead>
</table>

**FACTOR D: Inadequacy of Regulation**

<table>
<thead>
<tr>
<th>Country</th>
<th>Incidental Bycatch in Fishing Gear</th>
<th>Pollution and Oil Exploration</th>
<th>Climate Change</th>
<th>Natural Disasters</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>1</td>
<td>1</td>
<td>1</td>
<td>2</td>
</tr>
<tr>
<td></td>
<td>4, 5</td>
<td>3</td>
<td>1, 2, 3, 4, 5</td>
<td>1, 2, 4</td>
</tr>
</tbody>
</table>

- Guinea-Bissau: enforcement limited
- Ghana: continues despite strict laws
- Benin: not include green turtles
- Nigeria: TEDs not required
- Equatorial Guinea: organized harvest continues
- Demo of Congo: no commitment to law
- Guyana: license to take turtles
- Turks and Caicos: size limit for turtles

**FACTOR E: Other**

<table>
<thead>
<tr>
<th>Incidental Bycatch in Fishing Gear</th>
<th>Pollution and Oil Exploration</th>
<th>Climate Change</th>
<th>Natural Disasters</th>
</tr>
</thead>
<tbody>
<tr>
<td>1</td>
<td>1</td>
<td>1</td>
<td>2</td>
</tr>
<tr>
<td>4, 5</td>
<td>3</td>
<td>1, 2, 3, 4, 5</td>
<td>1, 2, 4</td>
</tr>
<tr>
<td>5, 6 (TED)</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>FACTOR A: Habitat</td>
<td>KNOWN THREATS</td>
<td>EXTENT</td>
<td>LIFE STAGE</td>
</tr>
<tr>
<td>-----------------------------------</td>
<td>------------------------------------------------------------------------------</td>
<td>-----------</td>
<td>-----------------------------------------------------------------------------</td>
</tr>
<tr>
<td>Erosion from storm events and sand extraction</td>
<td>1</td>
<td>1, 2, 3</td>
<td></td>
</tr>
<tr>
<td>Climate change: Sea level rise and increased storm events- loss of habitat</td>
<td>1</td>
<td>1, 2, 3</td>
<td></td>
</tr>
<tr>
<td>Dredging</td>
<td>2</td>
<td>4, 5</td>
<td></td>
</tr>
</tbody>
</table>

| FACTOR B: Overutilization        | Historic-intentional harvest                                                                 | 1         | 1, 2, 4, 5                                                                 | 4. Unknown                                                                                       |
|                                  | Current international - all countries participate, some, none                           | 2         | 1, 2, 4, 5                                                                 | 1. (Maldives, Mahe, Praslin, La Digue, Eparses Islands)                                           |

| FACTOR C: Disease                | FP                                                                                      | 2         | 4, 5                                                                        | 5. Threat to Population Stability                                                               |

| FACTOR C: Predation              | Beach                                                                                   | 1         | 2, 3                                                                        | 5. Threat to Population Stability                                                               |

| FACTOR D: Inadequacy of Regulation | Current international                                                                   | 4, 5      | 1. Present                                                                   | 1. Present                                                                                       |
|                                   | Country                                                                                 |           |                                                                             | 4. Threat to Population Stability                                                               |
|                                   | Local                                                                                   |           |                                                                             | 5. Unknown                                                                                       |

| FACTOR E: Other                  | Incidental Bycatch in Fishing Gear                                                      | 1         | 4, 5                                                                        | 5. Unknown                                                                                       |
|                                  | Climate Change                                                                          | 1         | 1, 2, 3, 4, 5                                                              | 3. Threat to Population Stability                                                               |
|                                  | Natural Disasters                                                                       | 2         | 1, 2, 4                                                                     | 3. Threat to Population Stability                                                               |
## Known Threats

<table>
<thead>
<tr>
<th>North Indian</th>
<th>KNOWN THREATS</th>
<th>LIFE STAGE</th>
<th>LEVEL</th>
</tr>
</thead>
</table>

### FACTOR A: Habitat

- Coastal Development with lighting: 2, 1, 2, 3, 3
- Erosion from storm events and sand extraction: 1, 2, 2
- Vehicles and pedestrian traffic on beach: 2, 1, 2, 3
- Beach and marine pollution: 2, ?, ?
- Climate change: trophic changes to foraging: 1, 2, 1, 2, 3
- Fishing practice: trawling: 1
- Dredging: 2

### FACTOR B: Overutilization

- Historic-intentional harvest: 1, 1, 2, 4, 5, 4
- Current international: 2, 1, 2, 4, 5, 1

### FACTOR C: Disease

- FP: Not known

### FACTOR C: Predation

- Beach: 1, 2, 3, 5

---

Comment [A11]: Comment: FACTOR A Vehicle and pedestrian traffic: the two specifically identified countries do not have green turtle nesting, so unless all DPS include problems at nesting beaches for any sea turtle species these indications should be removed (as mentioned previously in this review). Conversely, the text in Section 9 mentions vehicles on the beach and boats parked on beaches in Oman, but this country is not mentioned.

Response: Revised in text

Comment [A12]: Comment: Section 9 makes many references to trawling being widespread in this DPS and yet it is given no number in this table. Needs correction.

Response: Revised in text

Comment [A13]: Comment: Again, Oman is omitted. Section 9 clearly states "green turtles and sea turtle eggs are still frequently being harvested for food in Oman (R. Baldwin, Five Oceans LLC, pers. comm., 2013).". The same section mentions turtles killed in Eritrea and eggs taken in Iran, thus the inclusion of the 5 country names and not others is misleading.

Response: Revised in text

Comment [A14]: Comment: This section is so broad and vague as to have no concrete meaning and the text for Life stage does not seem appropriate. Consider serious revision.

Response: focus on FP as main disease for green turtles
<table>
<thead>
<tr>
<th>North Indian</th>
<th>Known threats</th>
<th>KNOWN THREATS</th>
<th>LIFE STAGE</th>
<th>LEVEL</th>
</tr>
</thead>
<tbody>
<tr>
<td>Current international</td>
<td>4, 5</td>
<td>1</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**FACTOR D: Inadequacy of Regulation**

<table>
<thead>
<tr>
<th>Country</th>
<th></th>
</tr>
</thead>
<tbody>
<tr>
<td>Djibouti- only recently more active, Somalia- insufficient and not enforced, Sudan- fishing law but not specific protection for sea turtles, Yemen- enforcement undefined</td>
<td></td>
</tr>
</tbody>
</table>

**FACTOR E: Other**

| Incidental Bycatch in Fishing Gear | 1 | 4, 5 | 5 |
| Vessel strikes | 2 | 4, 5 | 3 |
| Pollution | 3 | 1, 2, 3, 4, 5 | 3 |
| Climate Change | 1 | 1, 2, 3, 4, 5 | 3 |
| Natural Disasters | 2 | 1, 2, 4 | 3 |

*Comment [A15]: Comment: Pollution: Text in section 9 indicates cement pollution is a problem for nests in Saudi Arabia and light pollution is a problem at several locations, yet egg and hatchlings are not indicated as having a problem in this section.*

*Response: Added stages for lighting and cement beach*
<table>
<thead>
<tr>
<th><strong>Factor A: Habitat</strong></th>
<th><strong>Known Threats</strong></th>
<th><strong>Extent</strong></th>
<th><strong>Life Stage</strong></th>
<th><strong>Level</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>Coastal Development with lighting (oil flares off shore)</td>
<td>1</td>
<td>1, 2, 3</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>Erosion from storm events (tsunami) and sand mining and ports</td>
<td>2</td>
<td>1, 2</td>
<td>2</td>
<td></td>
</tr>
<tr>
<td>Vehicle and pedestrian traffic</td>
<td>2</td>
<td>2</td>
<td>2</td>
<td></td>
</tr>
<tr>
<td>Climate change: sea level rise</td>
<td>1</td>
<td>1, 2</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td>Pollution: siltation and degradation</td>
<td>1</td>
<td>3, 4, 5</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>Fishing practices including seagrass collection</td>
<td>2</td>
<td>3, 4, 5</td>
<td>5</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>Factor B: Overutilization</strong></th>
<th><strong>Known Threats</strong></th>
<th><strong>Extent</strong></th>
<th><strong>Life Stage</strong></th>
<th><strong>Level</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>Historic-intentional harvest</td>
<td>1</td>
<td>1, 2, 4, 5</td>
<td>4</td>
<td></td>
</tr>
<tr>
<td>Current Intentional harvest</td>
<td>2</td>
<td>1, 2, 4, 5</td>
<td>5</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>Factor C: Disease</strong></th>
<th><strong>Known Threats</strong></th>
<th><strong>Extent</strong></th>
<th><strong>Life Stage</strong></th>
<th><strong>Level</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>FP</td>
<td>1</td>
<td>4, 5</td>
<td>5</td>
<td></td>
</tr>
</tbody>
</table>

<table>
<thead>
<tr>
<th><strong>Factor C: Predation</strong></th>
<th><strong>Known Threats</strong></th>
<th><strong>Extent</strong></th>
<th><strong>Life Stage</strong></th>
<th><strong>Level</strong></th>
</tr>
</thead>
<tbody>
<tr>
<td>Beach and water</td>
<td>1</td>
<td>2, 3, 5</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>East Indian-West Pacific</td>
<td>KNOWN THREATS</td>
<td>EXTENT</td>
<td>LIFE STAGE</td>
<td>LEVEL</td>
</tr>
<tr>
<td>-------------------------</td>
<td>---------------</td>
<td>--------</td>
<td>------------</td>
<td>-------</td>
</tr>
<tr>
<td>FACTOR D: Inadequacy of Regulation</td>
<td>Current international</td>
<td>1</td>
<td>4, 5</td>
<td>1</td>
</tr>
<tr>
<td>Country</td>
<td>Australia, Japan, Andaman and Nicobar - local consumption, Vietnam - Destructive fishing illegal but still occurs. Myanmar, China and Vietnam - collection for trade and consumption occur even with ban, ATMR - Reserve - not demarcated, not include green turtles</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Local</td>
<td>5</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>FACTOR E: Other</td>
<td>Incidental Bycatch in Fishing Gear</td>
<td>1</td>
<td>4, 5</td>
<td>5, 6 (TEDs in some areas)</td>
</tr>
<tr>
<td></td>
<td>Pollution and debris</td>
<td>3</td>
<td>4, 5</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Climate Change</td>
<td>1</td>
<td>1, 2, 3, 4, 5</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Natural Disasters</td>
<td>2</td>
<td>1, 2, 4</td>
<td>3</td>
</tr>
</tbody>
</table>
### Central West Pacific

<table>
<thead>
<tr>
<th>KNOWN THREATS</th>
<th>EXTENT</th>
<th>LIFE STAGE</th>
<th>LEVEL</th>
</tr>
</thead>
<tbody>
<tr>
<td>protected beaches, 3. Portion of DPS on low density nesting beaches</td>
<td>(neritic and oceanic), 5. Adult (neritic and oceanic)</td>
<td>5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice</td>
<td></td>
</tr>
<tr>
<td></td>
<td></td>
<td>Minimizes in Some or All of DPS</td>
<td></td>
</tr>
</tbody>
</table>

**FACTOR A: Habitat**

- **Coastal Development (armoring) with lighting**
  - Extent: 1
  - Life Stage: 1, 2, 3
  - Level: 3

- **Erosion from storm events and sand extraction**
  - Extent: 1
  - Life Stage: 1, 2
  - Level: 3

- **Vehicle and pedestrian traffic**
  - Extent: 3
  - Life Stage: 1, 2, 3
  - Level: 3

- **Non-native vegetation**
  - Extent: 3
  - Life Stage: 1, 2
  - Level: 5

- **Pollution: beach and sedimentation and runoff**
  - Extent: 2
  - Life Stage: 1, 2, 3, 4, 5
  - Level: 5

- **Fishing practices (destructive)**
  - Extent: 2
  - Life Stage: 3, 4, 5
  - Level: 5

- **Dredging**
  - Extent: 2
  - Life Stage: 4, 5
  - Level: 5

**FACTOR B: Overutilization**

- **Historic-intentional harvest**
  - Extent: 1
  - Life Stage: 1, 2, 4, 5
  - Level: 4

- **Current Intentional harvest**
  - Extent: 1
  - Life Stage: 1, 2, 4, 5
  - Level: 4

**FACTOR C: Disease**

- **FP**
  - Extent: 2
  - Life Stage: 4, 5
  - Level: 5

**FACTOR C: Predation**

- **Beach**
  - Extent: 1
  - Life Stage: 1, 2, 3
  - Level: 5

**FACTOR D: Inadequacy of Regulation**

- **Current international- all countries participate, some, none**
  - Extent: not all
  - Life Stage: 4, 5
  - Level: 1

- **Micronesia- local consumption, Palau- only certain life stages**
  - Extent: Micronesia- local consumption, Palau- only certain life stages
  - Life Stage: 5

Comment [A16]: Comment: What animal predates nesting females on the beach? At least there is no predator for nesting females in Ogasawara islands.

Response: revised in text
<table>
<thead>
<tr>
<th>Central West Pacific</th>
<th>KNOWN THREATS</th>
<th>EXTENT</th>
<th>LIFE STAGE</th>
<th>LEVEL</th>
</tr>
</thead>
<tbody>
<tr>
<td>FACTOR E: Other</td>
<td>Incidental Bycatch in Fishing Gear</td>
<td>1</td>
<td>4, 5</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td>Pollution</td>
<td>3</td>
<td>4, 5</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Vessel strikes</td>
<td>3</td>
<td>4, 5</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td>Climate Change</td>
<td>1</td>
<td>1, 2, 3, 4, 5</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Natural Disasters</td>
<td>2</td>
<td>1, 2, 4</td>
<td>3</td>
</tr>
<tr>
<td>South Pacific</td>
<td>KNOWN THREATS</td>
<td>EXTENT</td>
<td>LIFE STAGE</td>
<td>LEVEL</td>
</tr>
<tr>
<td>---------------</td>
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<td>-------</td>
</tr>
</tbody>
</table>

| FACTOR A: Habitat | | | | |
| Coastal Development (armoring and erosion control structures) with lighting | 2 | 1, 2, 3 | 3 |
| Lighting from oil and gas | 3 | 2 | |
| Erosion from flooding and sand extraction | 1 | 1, 2 | 3 |
| Vehicle and pedestrian traffic | 2 | 1, 2, 3 | 3 |
| Non-native vegetation | 2 | 1, 2 | 5 |
| Pollution: beach and sedimentation and runoff | 2 | 1, 2, 3, 4, 5 | 5 |
| Fishing practices (destructive) | 2 | 3, 4, 5 | 5 |
| Climate change: sea level rise-atoll morphology | 2 | 1, 2 | 3 |
| Dredging | 2 | 4, 5 | 5 |

| FACTOR B: Overutilization | | | |
| Historic-intentional harvest | 2 | 1, 2, 4, 5 | 4 |
| Current Intentional harvest | 3 | 1, 2, 4, 5 | 2 |

| FACTOR C: Disease | | |
| FP | 2 | 4, 5 | 5 |

<p>| FACTOR C: Predation | | |
| Beach and water | 1 | 1, 2, 3, 4 | 5 |</p>
<table>
<thead>
<tr>
<th>Southwest Pacific</th>
<th>KNOWN THREATS</th>
<th>EXTENT</th>
<th>LIFE STAGE</th>
<th>LEVEL</th>
</tr>
</thead>
<tbody>
<tr>
<td>FACTOR D: Inadequacy of Regulation</td>
<td>Current international</td>
<td>4, 5</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Country</td>
<td>New Caledonia- take prohibited only during certain time period</td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Local</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>FACTOR E: Other</td>
<td>Incidental Bycatch in Fishing Gear</td>
<td>1, 4, 5</td>
<td>5, 6 (TEDs in Australia)</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Shark control programs</td>
<td>2, 4, 5</td>
<td>5, 6</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Vessel strikes, port dredging, military activities</td>
<td>2, 4, 5</td>
<td>3</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Toxic compounds and debris</td>
<td>2, 4, 5</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Climate Change- sea surface temp</td>
<td>1, 2, 3, 4, 5</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td></td>
<td>Natural Disasters</td>
<td>2, 1, 2</td>
<td>5</td>
<td></td>
</tr>
<tr>
<td>Central South Pacific</td>
<td>KNOWN THREATS</td>
<td>EXTENT</td>
<td>LIFE STAGE</td>
<td>LEVEL</td>
</tr>
<tr>
<td>----------------------</td>
<td>---------------</td>
<td>--------</td>
<td>------------</td>
<td>-------</td>
</tr>
<tr>
<td>FACTOR A: Habitat</td>
<td>Coastal development and associated lighting</td>
<td>3</td>
<td>1, 2, 3</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Erosion from sand mining</td>
<td>1</td>
<td>1, 2</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Pollution: beach and sedimentation and runoff</td>
<td>2</td>
<td>1, 2, 3, 4, 5</td>
<td>5</td>
</tr>
<tr>
<td></td>
<td>climate change: sea level</td>
<td>2</td>
<td>1, 2</td>
<td>3</td>
</tr>
<tr>
<td></td>
<td>Dredging</td>
<td>2</td>
<td>4, 5</td>
<td>5</td>
</tr>
<tr>
<td>FACTOR B: Overutilization</td>
<td>Historic-intentional harvest</td>
<td>2</td>
<td>1, 2, 4, 5</td>
<td>4</td>
</tr>
<tr>
<td></td>
<td>Current Intentional harvest</td>
<td>1</td>
<td>1, 2, 4, 5</td>
<td>2</td>
</tr>
<tr>
<td>FACTOR C: Disease</td>
<td>FP</td>
<td>0</td>
<td>0</td>
<td>0</td>
</tr>
<tr>
<td>FACTOR C: Predation</td>
<td>Beach</td>
<td>1</td>
<td>1, 2, 3</td>
<td>5, 6</td>
</tr>
<tr>
<td>FACTOR D: Inadequacy of</td>
<td>Current international</td>
<td>4, 5</td>
<td>1</td>
<td></td>
</tr>
<tr>
<td>Central South Pacific</td>
<td>KNOWN THREATS</td>
<td>EXTENT</td>
<td>LIFE STAGE</td>
<td>LEVEL</td>
</tr>
<tr>
<td>----------------------</td>
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</tr>
</tbody>
</table>

| Regulation | America Samoa- ESA but lack of enforcement, Tuvalu and French Polynesia effectiveness of ordinance not clear, Cook Islands and Pitcairn Islands traditional practices exception, Tokelau harvested prohibited but continues, Tonga certain size and season, Fiji inadequate compliance and enforcement | 5 |

<table>
<thead>
<tr>
<th>Country</th>
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FACTOR E: Other
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<td>climate change: sea level and storm events, trophic changes</td>
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<td>Current international - all countries participate, some, none</td>
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Conservation Efforts to protect all life stages of green turtles are effected at a local level. Local conservation efforts such as education and nest protection are growing. Most of the countries within each of the DPS have legislation protecting green turtles to varying degrees. The effectiveness of these protections is dependent on funding and commitment to enforcement.

Several international agreements provide legal protection for green turtles. The effectiveness of some of these international instruments also varies due to many factors such as participation, funding, and compliance (Table 16.8).
### Table 16.8. Summary of International instruments and the DPSs to which each applies:

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**Instruments**

- **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)**
  
- **Convention on the Conservation of Migratory Species of Wild Animals**
  
- **Convention on Biological Diversity**
  
- **Convention on International Trade in Endangered Species of Wild Fauna and Flora**
  
- **Convention on the Conservation of European Wildlife and Natural Habitats**
  
- **Convention for the Co-operation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region**

**Comment [A17]:** Comment: Can you double check that none of the countries in Central South Pacific DPS belong to Forum Fisheries Authority (FFA), or Western/Central Pacific Fisheries Commission (WCPFC)?

**Response:** Revised to correct-WCPFC was removed since it was duplicative of the Convention for the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean.
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<td>(Abidjan Convention)</td>
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Comment [A18]: Comment: #7 and #8, as well? 
Response: checked and revised

Comment [A19]: Comment: #6, as well? 
Response: Checked and revised
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International Convention for the Presentation of Pollution from Ships (MARPOL)

International union for conservation of nature

Inter-American Tropical Tuna Commission (IATTC)

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 ASEAN Sea Turtle Conservation and Protection

Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa (Abidjan Memorandum).


Comment [A20]: Comment:: #9 and #10 as well?
Response: Revised in text
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</thead>
<tbody>
<tr>
<td>Protocol Concerning Specially Protected Areas and Biological Diversity in the Mediterranean</td>
<td>✓</td>
<td></td>
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<td></td>
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</tr>
<tr>
<td>Ramsar Convention on Wetlands</td>
<td>✓</td>
<td>✓</td>
<td></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>Secretariat of the Pacific Regional Environment Programmed (SPREP)</td>
<td>✓</td>
<td>✓</td>
<td></td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td></td>
<td></td>
<td></td>
<td></td>
</tr>
<tr>
<td>South-East Atlantic Fisheries Organization (SEAFO)</td>
<td>✓</td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
<td></td>
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</tr>
<tr>
<td>Torres Strait Treaty of 1978</td>
<td></td>
<td></td>
<td></td>
<td></td>
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<td></td>
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<td></td>
</tr>
<tr>
<td>United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td></td>
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<td></td>
<td></td>
</tr>
<tr>
<td>United States Magnuson-Stevens Fishery Conservation and Management Act</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td>✓</td>
<td></td>
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<td></td>
</tr>
</tbody>
</table>
16.3. Extinction Risk

The SRT assessed the extinction risk for green turtles in each DPS, focusing on the six critical assessment elements (Table 16.9). Each SRT voting member ranked the importance of each of the population elements (first four above) by assigning them a value from 1 to 5, with 1 representing a very low risk. They ranked the influence of the five factors (threats) on the status of the DPS by assigning a value of 0 (neutral) to −2, and ranked the influence of conservation efforts on the status of the DPS by assigning a value of 0 to 2. The SRT noted that none of these elements is entirely independent, and did not attempt to use the values applied to each element by each SRT member to arrive at extinction risk.

Table 16.9. SRT voting result for each of the six critical assessment elements.

<table>
<thead>
<tr>
<th></th>
<th>Element 1</th>
<th>Element 2</th>
<th>Element 3</th>
<th>Element 4</th>
<th>Element 5</th>
<th>Element 6</th>
</tr>
</thead>
<tbody>
<tr>
<td>DPS 1 - N Atl</td>
<td>1.18</td>
<td>1.18</td>
<td>1.45</td>
<td>1.36</td>
<td>-0.45</td>
<td>0.82</td>
</tr>
<tr>
<td>DPS 2 - Med</td>
<td>3.92</td>
<td>2.75</td>
<td>3.58</td>
<td>3.08</td>
<td>-1.25</td>
<td>0.5</td>
</tr>
<tr>
<td>DPS 3 - S Atl</td>
<td>1.58</td>
<td>1.92</td>
<td>1.33</td>
<td>1.67</td>
<td>-0.83</td>
<td>0.75</td>
</tr>
<tr>
<td>DPS 4 - SW Ind</td>
<td>1.25</td>
<td>1.75</td>
<td>1.42</td>
<td>1.58</td>
<td>-0.75</td>
<td>0.75</td>
</tr>
<tr>
<td>DPS 5 - N Ind</td>
<td>1.42</td>
<td>2</td>
<td>1.58</td>
<td>2</td>
<td>-0.92</td>
<td>0.33</td>
</tr>
<tr>
<td>DPS 6 - E Ind/W Pac</td>
<td>1.67</td>
<td>3.08</td>
<td>1.67</td>
<td>1.5</td>
<td>-1.5</td>
<td>0.5</td>
</tr>
<tr>
<td>DPS 7 - CW Pac</td>
<td>2.5</td>
<td>2.42</td>
<td>2.17</td>
<td>2.17</td>
<td>-1.08</td>
<td>0.67</td>
</tr>
<tr>
<td>DPS 8 - SW Pac</td>
<td>1.17</td>
<td>1.67</td>
<td>1.5</td>
<td>1.42</td>
<td>-0.67</td>
<td>0.58</td>
</tr>
<tr>
<td>DPS 9 - CS Pac</td>
<td>3.18</td>
<td>2.91</td>
<td>1.91</td>
<td>2.18</td>
<td>-1.27</td>
<td>0.55</td>
</tr>
<tr>
<td>DPS 10 - CN Pac</td>
<td>2.67</td>
<td>1.33</td>
<td>3</td>
<td>2.58</td>
<td>-0.92</td>
<td>0.5</td>
</tr>
<tr>
<td>DPS 11 - E Pac</td>
<td>1.64</td>
<td>1.18</td>
<td>1.73</td>
<td>1.64</td>
<td>-0.91</td>
<td>1.09</td>
</tr>
</tbody>
</table>
The SRT then assessed the probability that each DPS will reach a critical risk threshold within 100 years, throughout all or a significant portion of its range. Each SRT member assigned 100 points across the rank categories (Table 16.10 and Table 16.11).

### Table 16.10. Summary of risk threshold scores for each DPS.

<table>
<thead>
<tr>
<th>Percent Range</th>
<th>DPS 1 North Atlantic</th>
<th>DPS 2 Mediterranean</th>
<th>DPS 3 SW Atlantic</th>
<th>DPS 4 SW Indian</th>
<th>DPS 5 North Indian</th>
<th>DPS 6 East Indian/West Pacific</th>
<th>DPS 7 CW Pacific</th>
<th>DPS 8 SW Pacific</th>
<th>DPS 9 Pacific</th>
<th>DPS 10 Pacific</th>
<th>DPS 11 Pacific</th>
</tr>
</thead>
<tbody>
<tr>
<td>&lt;1%</td>
<td>87.0</td>
<td>10.1</td>
<td>69.0</td>
<td>71.3</td>
<td>68.4</td>
<td>47.5</td>
<td>43.3</td>
<td>72.5</td>
<td>38.2</td>
<td>47.8</td>
<td>63.6</td>
</tr>
<tr>
<td>1-5%</td>
<td>3.0</td>
<td>11.8</td>
<td>16.5</td>
<td>13.6</td>
<td>19.8</td>
<td>17.3</td>
<td>19.3</td>
<td>9.1</td>
<td>19.9</td>
<td>15.9</td>
<td>16.7</td>
</tr>
<tr>
<td>6-10%</td>
<td>1.4</td>
<td>17.6</td>
<td>9.9</td>
<td>7.6</td>
<td>9.7</td>
<td>11.2</td>
<td>13.8</td>
<td>10.6</td>
<td>20.5</td>
<td>14.1</td>
<td>6.4</td>
</tr>
<tr>
<td>11-20%</td>
<td>4.1</td>
<td>27.9</td>
<td>4.2</td>
<td>5.5</td>
<td>1.7</td>
<td>9.4</td>
<td>12.5</td>
<td>5.4</td>
<td>8.2</td>
<td>7.5</td>
<td>7.8</td>
</tr>
<tr>
<td>21-50%</td>
<td>4.1</td>
<td>23.9</td>
<td>0.4</td>
<td>2.0</td>
<td>0.5</td>
<td>7.1</td>
<td>8.6</td>
<td>2.4</td>
<td>9.5</td>
<td>9.8</td>
<td>5.0</td>
</tr>
<tr>
<td>&gt;50%</td>
<td>0.5</td>
<td>8.8</td>
<td>0.0</td>
<td>0.0</td>
<td>0.0</td>
<td>7.5</td>
<td>2.7</td>
<td>0.0</td>
<td>3.7</td>
<td>4.9</td>
<td>0.5</td>
</tr>
</tbody>
</table>

The table above presents the summary of risk threshold scores for each DPS, categorized by percentage ranges.
Table Figure 16.111. Bar graph of risk threshold scores.

Comment [A21]: MAKE SURE CORRECTED IN TOC

Comment [A22]: Comment: This is not a table (!) and does not add further information over Table 16.10 and hence should be dropped.

Response: It summarizes the distribution of votes for each DPS.