NOAA Technical Memorandum NMFS

MARCH 2015

STATUS REVIEW OF THE GREEN TURTLE *(CHELONIA MYDAS)* UNDER THE ENDANGERED SPECIES ACT

Jeffrey A. Seminoff, Camryn D. Allen, George H. Balazs, Peter H. Dutton, Tomoharu Eguchi, Heather L. Haas, Stacy A. Hargrove, Michael Jensen, Dennis L. Klemm, Ann Marie Lauritsen, Sandra L. MacPherson, Patrick Opay, Earl E. Possardt, Susan Pultz, Erin Seney, Kyle S. Van Houtan, Robin S. Waples

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U. S. DEPARTMENT OF COMMERCE
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Southwest Fisheries Science Center

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PREFACE

The Green Turtle Status Review Team (SRT) has undertaken a review consistent with section 4(a)(1) of the Endangered Species Act (ESA), using the best available scientific information. The SRT assessed the green turtle population structure globally in order to determine whether the green turtle could be listed as one or more Distinct Population Segments (DPSs). After determining that DPSs could be identified and identifying potential DPSs, the SRT assessed the risk of extinction for each potential DPS using a structured decision-making process that combined analysis of large amounts of empirical data with expert opinion. Extinction risk probabilities were considered based on six different critical assessment elements, including four indicators of "Viable Turtle Populations" (abundance, productivity, spatial structure, diversity / resiliency), as well as an assessment of threats and conservation efforts.

The SRT report drew conclusions for each potential DPS regarding extinction risk (using quasi-extinction thresholds) under current management regimes. In doing this, the SRT considered the six critical assessment elements listed above based on the assumption that ESA protective measures would continue into the future. The SRT was not asked to, and did not, speculate on extinction risk under a theoretical scenario in which the green turtle was not listed under the ESA. The SRT also was not asked to assess the role of the ESA in the conservation of the species following listing. The SRT did not make any listing recommendations regarding status under the ESA; those listing determinations will be made separately by a management team. When making its listing determination, the management team will rely on the SRT analysis and report as well as any other management considerations and additional information, including its assessment of increased risk to the species due to inadequacy of regulatory mechanisms under a scenario without ESA protections where they apply.

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 (USFWS); 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 USFWS 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, 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. A portion of the range of a species is considered to be a significant portion of its range (SPR) if it's contribution to the viability of the species is so important that, if green turtles were extirpated within it, the remaining portion of the population would be in danger of extinction. Only two DPSs were found to have potential SPRs, the Central North Pacific DPS, and the East Indian-West Pacific DPS.

Finally, each SRT voting member gave their expert opinion on the likelihood that each DPS would reach a critical risk threshold (quasi-extinction) within 100 years by spreading 100 points across several risk categories for each DPS. For DPSs that were determined to have potential SPRs, the SRT conducted two votes for the risk of extinction: 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 extinction, and conclusions on SPR for each DPS is found in the DPS-specific sections (Sections 5–15) of this report.

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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
CITES	Convention on the International Trade in Endangered Species of Wild Fauna and Flora
cm	centimeter
CMAR	ETP Marine Corridor Initiative
CMS	Convention on the Conservation of Migratory Species of Wild Animals
COFI	FAO Committee on Fisheries
DNA	Deoxyribonucleic Acid
DPS	Distinct Population Segment
EEZ	Exclusive Economic Zone
ENSO	El Niño Southern Oscillation
ESA	Endangered Species Act of 1973, as amended
ESU	Evolutionary Significant Unit
ETP	Eastern Tropical Pacific
FAO	Food and Agriculture Organization of the United Nations
FP	Fibropapillomatosis
FFS	French Frigate Shoals
FWC	Florida Fish and Wildlife Conservation Commission
FST	Genetics Fixation Index
GBR	Great Barrier Reef
GIWA	Global International Waters Assessment
km	kilometer
IAC	Inter-American Convention for the Protection and Conservation of Marine Turtles
IATTC	InterAmerican Tropical Tuna Commission
ICCAT	International Commission for the Conservation of Atlantic Tunas
IOSEA	Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding
IOTC	Indian Ocean Tuna Commission
IPCC	Intergovernmental Panel on Climate Change
IRG	In-water Research Group
IUCN	International Union for the Conservation of Nature
km	kilometer
l	liter
LME	Large Marine Ecosystem
m	meter

MARPOL	International Convention for the Prevention of Pollution from Ships
mi	mile
ml	milliliter
MNS	Mean Nesting Size
MOU	Memorandum of Understanding
MPA	Marine Protected Area
MSA	U.S. Magnuson-Stevens Fishery Conservation and Management Act
mya	million years ago
MHI	Main Hawaiian Islands
mtDNA	mitochondrial Deoxyribonucleic Acid
MX	Mexico
nDNA	nuclear Deoxyribonucleic Acid
nGBR	northern Great Barrier Reef
NGO	Non-governmental organization
NMFS	National Marine Fisheries Service
NPF	Northern Australian Prawn Fishery
NRC	National Research Council
NWHI	Northwest Hawaiian Islands
PIT	Passive Integrated Transponder
PVA	Population Viability Analysis
QET	Quasi-extinction Threshold
RI	Remigration Interval
RFMO	Regional Fishery Management Organization
SCL	Straight Carapace Length
SD	Standard Deviation
SEAFO	South-East Atlantic Fisheries Organization
sGBR	southern Great Barrier Reef
SNP	single nucleotide polymorphism
SPAW	Protocol Concerning Specially Protected Areas and Wildlife
SPR	Significant Portion of its Range
SPREP	South Pacific Regional Environment Programme
SWIO	Southwest Indian Ocean
SQE	Susceptibility to Quasi-extinction
SRT	Status Review Team
TED	Turtle Excluder Device
TIHPA	Turtle Island Heritage Protection Area
USFWS	U.S. Fish and Wildlife Service
UK	United Kingdom
UNCLOS	United Nations Convention on the Law of the Sea
UNEP	United Nations Environment Programme
UNESCO	United Nations Educational, Scientific, and Cultural Organization
yr(s)	year(s)

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 (USFWS; 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. USFWS 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 USFWS 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 USFWS 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, USFWS 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.

USFWS 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 USFWS 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 USFWS 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 USFWS Field and Regional Offices, and NMFS and USFWS Headquarters Offices, and provided a diverse range of expertise, including green turtle population structure, biology, demography, ecology, and management challenges, as well as 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 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).

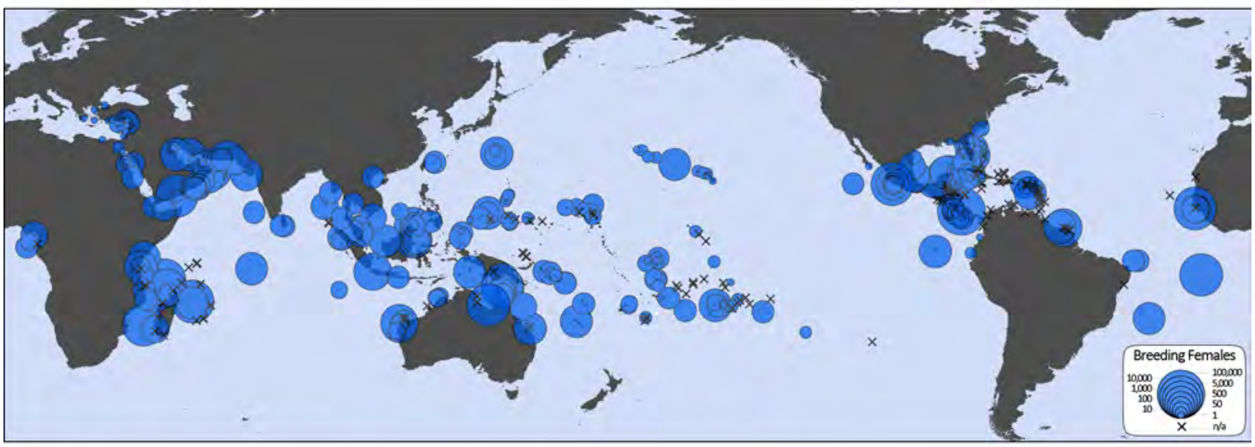

Figure 2.1. Nesting distribution of green turtles around the globe. Sites marked with an 'X' indicate known nesting sites, but for which no recent nesting abundance data are available.

The 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 (Broderick *et al*., 2006), Commonwealth of Australia (Australia; eastern: Limpus, 2009; western: Prince, 2000), Comoros Islands (Frazier, 1985; Bourjea *et al.*, 2007a; Innocenzi *et al.*, 2010), Eparses Islands (Tromelin Island and Europa Island: Lauret-Stepler *et al*., 2007), Federative Republic of Brazil (Brazil; Trindade Island: Almeida *et al*., 2011), Kingdom of Saudi Arabia (Saudi Arabia; Al-Merghani *et al*., 2000; PERSGA/GEF 2004), Malaysia (Chan, 2006, 2010; Basintal, 2011), Republic of Costa Rica (Costa Rica; Pacific Coast: Blanco *et al.*, 2012; Caribbean coast: (Bjorndal *et al.*, 1999; Chaloupka *et al.*, 2008), Republic of Ecuador (Ecuador; Galapagos Archipelago: Zárate *et al.,* 2006), Republic of Guinea-Bissau (Guinea-Bissau; Bijagos Archipelago: Barbosa *et al.*, 1998), Republic of Indonesia (Indonesia; Dethmers, 2010; Reischig *et al*., 2012), Republic of the Philippines (Burton, 2012), Republic of Seychelles (Seychelles; Mortimer *et al*., 2012), Republic of Suriname (Suriname; Turny, 2012), Sultanate of Oman

(Oman; Al Kindi *et al*., 2008), United Mexican States (Mexico; Yucatán Peninsula: Lopez, 2001; Xavier *et al.*, 2006; Michoacán: Delgado-Trejo and Alvarado-Figueroa, 2012), and United States of America (United States; Florida: Chaloupka *et al.*, 2008; Hawaii: Balazs and Chaloupka, 2004a; Chaloupka *et al.*, 2008).

Lesser nesting areas are located in Atoll of Manuae (Scilly Atoll; Lebeau, 1985), Bolivarian Republic of Venezuela (Venezuela; Prieto *et al.,* 2012; Vera and Buitrago, 2012), Chagos Archipelago (Mortimer and Day, 1999), Cook Islands (Palmerston Atoll; White 2012), Co-operative Republic of Guyana (Guyana; Pritchard, 1969), Commonwealth of Australia (Australia; Gulf of Carpentaria; Limpus, 2009), Democratic Republic of Yemen (Yemen; PERSA/GEF, 2004), Democratic Socialist Republic of Sri Lanka (Sri Lanka; Dattatri and Samarajiva, 1983; Kapurusinghe, 2006; Ekanayake *et al.*, 2011), Dominican Republic (Ottenwalder, 1981), d'Entrecasteaux Islands (Pritchard, 1994), Federative Republic of Brazil (Brazil; Atoll da Rocas: Bellini *et al.*, 2012), Federal Republic of Somalia (Somalia; Goodwin, 1971), Guiana (French Guiana; Fretey, 1984), Independent State of Papua New Guinea (Papua New Guinea; Kinch, 2003; Wangunu *et al.,* 2004), Islamic Republic of Iran (Iran; Mobaraki, 2004), Islamic Republic of Pakistan (Pakistan; Kabraji and Firdous, 1984), Japan (Kamezaki *et al.*, 2004), Kingdom of Thailand (Thailand; Groombridge and Luxmoore, 1989), Mayotte Archipelago (Bourjea *et al.*, 2007), Micronesia (Wetherall *et al.*, 1993), Natuna Islands (Limpus, 2009), New Caledonia (Limpus, 1985, 2009), People's Republic of Bangladesh (Bangladesh; Khan, 1982), People's Republic of China (China; Groombridge and Luxmoore, 1989), Primieras Islands (Hughes, 1974), Republic of Angola (Angola; Carr and Carr, 1991), Republic of Equatorial Guinea (Equatorial Guniea; Bioko Island: Tomás *et al.*, 1999), Republic of Ghana (Ghana; Fretey, 2001), Republic of Cuba (Cuba; Blanco *et al.*, 2009), 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 Marshall Islands (Marshall Islands; Bikar Atoll; McCoy, 2004), 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; Bureau of Marine Resources, 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; Lwin, 2009), Sao Tome é Principe (Brongersma, 1982), Socialist Republic of Vietnam (Vietnam; Hien, 2002) Solomon Islands (Leary and Laumani, 1989), United Republic of Tanzania (Tanzania; Howell and Mbindo, 1996), and United Mexican States (Mexico; Revillagigedos Islands: Holroyd and Telfry, 2010), 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, cold-stunning events at temperate foraging areas (Witherington and Ehrhart, 1989a; McMichael *et al.*, 2006) and, to a lesser extent, biointoxication from harmful algal blooms (i.e., red tides; e.g., Mendoza-Salgado *et al.*, 2003; Buss and Bengis, 2012) may lead to mortality of juvenile and adult green turtles, thereby impacting a population's present and future reproductive status.

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; Ackerman, 1997; Witherington, 1997; Lorne and Salmon, 2007). 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 hatchling turtles. Nests laid in these areas are at a higher risk due to tidal inundation (Schroeder and Mosier, 2000).

Global climate change has a substantial likelihood of altering habitat conditions in the future. Both the marine and terrestrial realm will be influenced by this phenomenon and will likely undergo alterations that will adversely impact green turtles. For example, sea level rise that is caused by climate change will lead to increased erosion of nesting beaches and significant loss of habitat (Baker *et al.*, 2006; IPCC, 2007); the extent to which green turtles can adapt to these spatial changes in nesting beach location and quality is unknown. Climate change will likely also cause higher sand temperatures leading to increased feminization of surviving hatchings (i.e. changes in sex ratio); some beaches will also experience lethal incubation temperatures that will result in losses of complete hatchling cohorts (Glen and Mrosovsky, 2004; Fuentes *et al.*, 2010b, 2011). Increased sea surface temperatures may alter the timing of nesting for some stocks (Weishampel *et al.*, 2004), although the implications of changes in nesting timing are unclear. Changes in sea temperatures will also likely alter seagrass, macroalgae, and invertebrate populations in coastal habitats in many regions (Scavia *et al.*, 2002).

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 ^{o}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 varies with population, ranging from approximately 28.0–30.3 ^{o}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 visual cues, orienting to the brightest horizon, which is over the ocean on natural 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 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, internesting habitat, and migratory habitat for adult green turtles (Plotkin, 2003; NMFS and USFWS, 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 zone by green turtles, much remains to be learned about how oceanography affects juvenile and adult survival, adult migration, prey availability, and reproductive output.

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.*, 2002c; Balazs and Chaloupka, 2004b). 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, 2004b; Kubis *et al.*, 2009; Patrício *et al.*, 2014).

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 (sGBR) 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 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 sGBR, 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 northen 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).

2.5.5. Green turtle life history and human impacts

Green turtles are generally regarded as a species of conservation concern and in many places throughout world they are impacted by a variety of anthropogenic threats to multiple life stages, including eggs, hatchlings, oceanic and neritic juveniles, and adults. Because green turtles spend greater than 99 percent of their lives in the sea, in-water impacts can have substantial negative affects on population viability. Because migratory routes of green turtles commonly cross territorial waters of many nations or occur in the high seas, efforts to protect this species can be complex and logistically challenging. With regard to the terrestrial realm, although green turtles spend only a small portion of their lives on beaches for nesting or basking, the fact that they depart the sea at specific and predictable areas make them vulnerable to point-source anthropogenic impacts.

Despite the fact that sea turtles have been the focus of research and conservation efforts for several decades in various places around the world (Frazier 2003), there are still very large gaps in our understanding of green turtle life history and demography. These gaps likely owe to logistical challenges of studying sea turtles when they are dispersed in the open ocean and to the long time spans from hatchling to maturity. However, even as our knowledge about green turtle biology increases, as a long-lived and slow-maturing species, the traits that make green turtles so vulnerable to reduced survival rates also make them very slow to recover once depleted, leaving them vulnerable to other threats even if the impact that initially caused their depletion is addressed (see Congdon *et al.,* 1993).

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.

Nesting Site	Mean Nesting Size (SCL, cm)	Remigration Interval (yrs.)	Nesting Frequency (nests/yr)	Clutch Size (eggs/clutch)	Survival Rates	Growth Rates (SCL; cm/year)	Age at First Reproduction (years)
NORTH ATLANTIC							
Archie Carr National Wildlife Refuge, Florida, USA	101.5	2	3	136	26–75% hatchling	Adult=F: 0.5; hatchling: 1.18-4.55	18–27
Core Index Beaches, FL, USA	–	–	10 max	128	–	–	–
El Cuyo, Yucatan, Mexico	–	–	2.73	129.7	–	–	–
Isla Holbox, Quintana Roo, Mexico	–	2-3	–	113.3	–	–	12–20, mean 16
Central Coast, Quintana Roo, Mexico	–	2-3	–	116	Hatchling= 81.98%	–	12–20
Isla Aguada, Campeche, Mexico	–	–	3.54–4.01	112.25	Hatchling= 55.8-61.7%	–	–
Tortuguero, Costa Rica	100.1	2.95	2.8	108	Adult= 55-82%	–	12-26
MEDITERRANEAN							
Akyantan, Turkey	–	–	–	108	–	–	–
Kazanli, Turkey	96	–	–	115	–	–	–
Samadang, Turkey	–	–	–	125	–	–	–
Alagadi, Cyprus	92 (CCL)	3	3	115	–	0.11 (CCL)	–

Nesting Site	Mean Nesting Size (SCL, cm)	Remigration Interval (yrs.)	Nesting Frequency (nests/yr)	Clutch Size (eggs/clutch)	Survival Rates	Growth Rates (SCL; cm/year)	Age at First Reproduction (years)
West Coast, Cyprus	–	3	4–5	100	–	–	–
Israel	–	3	–	105	–	–	–
Lattakia Beach, Syrian Arab Republic	–	–	–	108	–	–	–
Ras al-Bassit, Syrain Arab Republic	–	–	–	108	–	–	–
Wadi Kandil Beach, Syrian Arab Republic	–	–	–	108	–	–	–
SOUTH ATLANTIC							
Bioko Island, Equatorial Guinea	–	–	3	104.6–112.4	–	–	–
Poilão Bijagos Archipelago, Guinea Busseau	–	–	–	122–131.2	–	–	–
Ascension Island, UK	116.8	4	6	102.9	–	–	17–35
Aves Island, Venezuela	107.7	2-3	1.6–2.6	122.9–124	–	–	–
Galibi Reserve, Suriname	109	2-3	3.5	102–138	–	–	24-36
Isla Trindade, Brazil	115.2 (CCL)	3	~3–6,	120.1	–	–	–
Atol De Rocas, Brazil	115.9 (CCL)	3.5	4.3–5.2	121.2	–	–	–
SOUTHWEST INDIAN OCEAN							
Aldabra, Seychelles Islands	103	–	3	90-200	–	–	–
Mohéli, Comoros Islands	112.3 (CCL)	–	–	116	–	–	–

Nesting Site	Mean Nesting Size (SCL, cm)	Remigration Interval (yrs.)	Nesting Frequency (nests/yr)	Clutch Size (eggs/clutch)	Survival Rates	Growth Rates (SCL; cm/year)	Age at First Reproduction (years)
Mayotte, Comoros Islands, France	110.8 (CCL)	3	3.03	121.6	–	–	–
Tromelin, Esparces Islands, France	104.1	3	3	124.6–129	–	–	–
Europa, Esparces Islands, France	108.9	3	3	142–152	–	–	–
NORTH INDIAN OCEAN							
Gujarat, India	–	–	–	92.6	–	–	–
Hawkes Bay and Sandpit, Pakistan	–	–	–	108.5	–	–	–
Sharma, Peoples Democratic Republic of Yemen	96	–	–	106–122.4	–	–	–
Ras al Hadd, Oman	97.1	–	–	110.6	–	–	–
Ras Baridi, Saudi Arabia	105.2 (CCL)	–	–	103	–	–	–
Karan and Jana Islands, Arabian Gulf, Saudi Arabia	98 (CCL)	–	4	88.5	–	–	–
EAST INDIAN/WEST PACIFIC							
Sarawak, Malaysia	–	3	~5	104.7	–	–	–
Redang Island, Malaysia	–	4	6	100	–	–	–
Sipadan, Sabah, Malaysia	–	–	–	–	–	–	–

Nesting Site	Mean Nesting Size (SCL, cm)	Remigration Interval (yrs.)	Nesting Frequency (nests/yr)	Clutch Size (eggs/clutch)	Survival Rates	Growth Rates (SCL; cm/year)	Age at First Reproduction (years)
Sabah Turtle Islands, Malaysia	–	2.39	2.7	87.3	–	Nesting females=0.8 Juv=3.6	–
Berau Islands, Berawan Archipelago, Indonesia	–	2.9	4.38	91.2–99.0	–	–	25–30
Enu Island (Aru Islands), Indonesia	–	–	2-8	106	–	–	–
Vietnam	–	≥2–3	3-5	115	80% (artificial hatcheries)	–	–
Turtle Islands, Philippines	99.5 (CCL)	2.5	5	95.61	1% hatchling	–	–
Lanyu, Taiwan, Provence of China	–	4.3	2.8	105.5	–	–	–
Wan-an, Taiwan, Provence of China	–	4.6	3.2	104.6	–	–	–
Thameehla Island, Myanmar	–	1–4	–	93	–	–	–
Pangumbahan, Java, Indonesia	–	–	–	107	–	–	–
Sukamade, Java, Indonesia	–	–	2-4	113	–	–	–
Western Australia	–	2–5	2.93	–	–	–	–
Sri Lanka	–	2.5–3.5	4	112.1	–	–	–

Nesting Site	Mean Nesting Size (SCL, cm)	Remigration Interval (yrs.)	Nesting Frequency (nests/yr)	Clutch Size (eggs/clutch)	Survival Rates	Growth Rates (SCL; cm/year)	Age at First Reproduction (years)
CENTRAL WEST PACIFIC							
Ogasawara Islands, Japan	–	3.7	4.1	102	–	–	–
SOUTHWEST PACIFIC							
Heron Island, sGBR, Austraila	106 (CCL)	5.78	5.06	112.0–115.2	Adults=95% Juv=88%	0.6–2.1 (CCL)	40
Raine Island, nGBR, Australia	109 (CCL)	5.35	–	103.9	6.7–99.3%	1.2–4.1	25
CENTRAL SOUTH PACIFIC							
Rose Atoll, American Samoa	94.7	–	–	–	–	–	–

CENTRAL NORTH PACIFIC							
French Frigate Shoals, USA	92.2	3-4	4	104	–	1.5–2.5	35–40
EASTERN PACIFIC							
Michoacán, Mexico	82 (CCL)	1.8–3	3.1	65.1	Juv=58%; Adult=85–97%	0.72	9–47
Galapagos Islands, Ecuador	81.3	3	1.37	82.9	–	0.11–1.57	–

3. APPROACH TO STATUS REVIEW

3.1. Determination of Distinct Population Segments (DPSs)

The 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 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/USFWS policy defines a population to be a DPS if it is both discrete and significant relative to the taxon to which it belongs (USFWS 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 an 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 discrete 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 non-existent, 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: Adult Female Abundance = (total counted females / years of monitoring) x remigration interval. Nester abundance distribution is also presented in 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.). These depict 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 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

The next step in the process was assessment of extinction risk or, in this case, quasi-extinction risk—which the team called the critical risk threshold. Elaboration of quasi-estinction risk is a common practice in wildlife status assessments. We note that this is a different threshold than that for endangered or threatened under the ESA. Whereas quasi-extinction indicates that a population has reached the point of no return, threatened and endangered populations still have recovery potential. In order to assess quasi-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. To develop this process, we 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 (Good *et al.*, 2005), 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 defined quasi-extinction as reaching a state within 100 years at which the species (or DPS) 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 potential for recovery. We recognize that we assessed quasi-extinction risk within a relatively short timeframe (100 yrs) for a species that has a generation time between 30 and 40 years (generation time = age at first reproduction + ½ of reproductive longevity; Pianka, 1974), and that this may underestimate actual extinction risk. It's important to note that this process used was not designed to determine status under the ESA, such as endangered, threatened or not warranted.

Further, quasi extinction was assessed under current management regimes. The SRT did not assess how extinction risk would change if current protections under the ESA or other laws or international mechanisms, were removed or diminished. Yet this species is conservation dependent, i.e., nesting females and eggs are, and will always be, vulnerable on beaches; juveniles and adults will always be vulnerable to injurious or even fatal to interactions with fisheries; and harvest of both adults and eggs for human consumption and trade will remain without continued management. Extinction risk would likely be higher if management were removed.

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 / 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 McElhaney *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, 2005a, 2010, 2011; Conant *et al.*, 2009). 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, the votes for each element were essentially used to inform the voting on extinction risk and as an indicator of which weighed more heavily in the overall extinction risk. We 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 a critical risk threshold (quasi-extinction) within 100 years. For purposes of this exercise, the SRT agreed to define critical risk threshold (quasi-extinction) 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, 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 4 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[1]; (3) the range of a species is considered to be the general geographical area within which that species can be found at the time USFWS 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

[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.

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 at risk, 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 SRT's function was to determine potential DPS structure and identify extinction risk, as described above. The status review report 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). This will be reflected in a proposed rule to make these changes to the current listings of the green turtle.

4. DETERMINATION OF DPS

4.1. Overview of Information Used to Determine DPS

As noted in Section 1.1.4, joint NMFS/USFWS 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 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).

4.1.1.1. Genetics

The green turtle is present in all tropical and temperate ocean basins and has a life history that involves nesting on 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 6 and 17 mya; Rögl, 1998), and possibly coincided with the closing of the Isthmus of Panama (between 2.5 and 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 2.5 and 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 appear 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 percent (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).

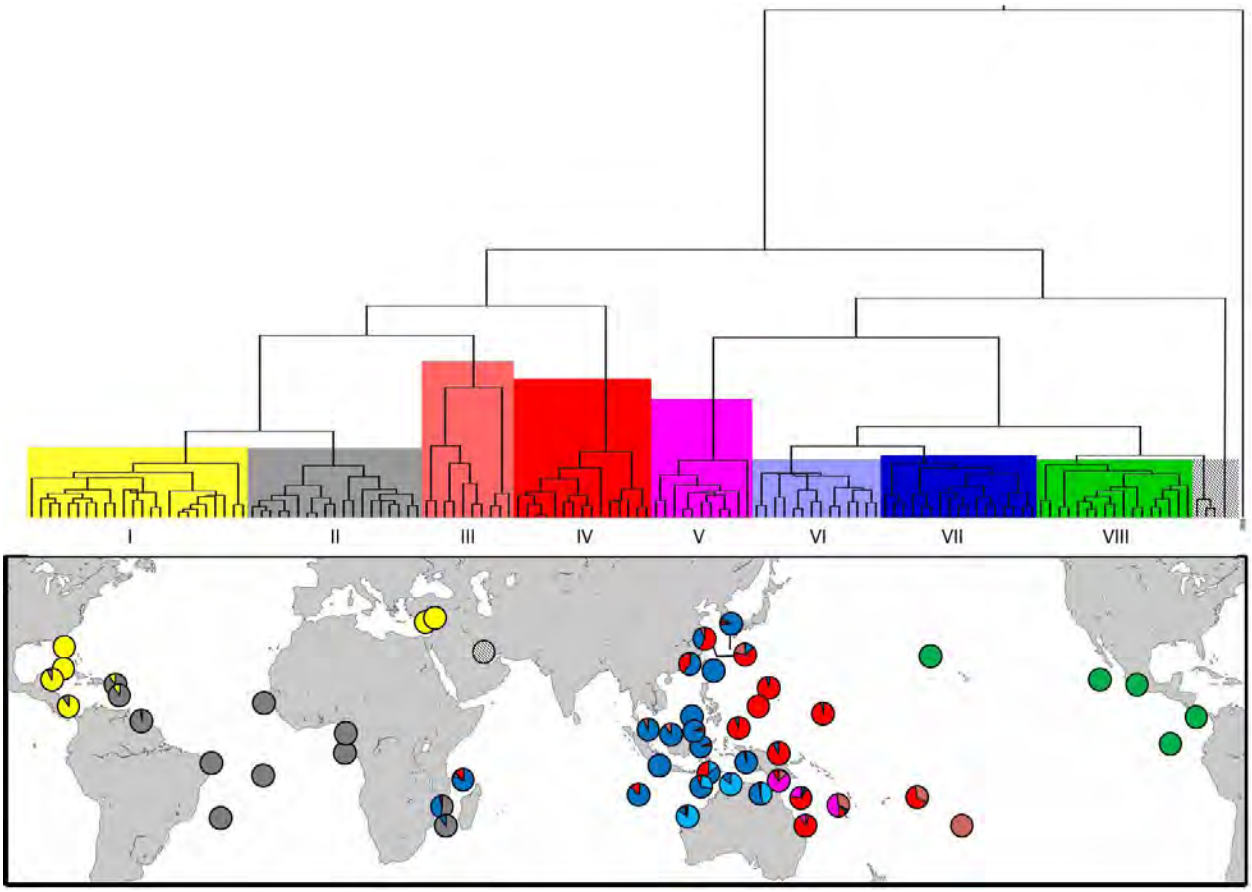

Figure 4.1. Bayesian phylogenetic tree showing relationships (average number of base substitutions) among 129 mtDNA haplotypes that group into eight major clades, each defined by a unique color. The flatback turtle, *Natator depressus* is used as an outgroup. The geographic distribution of haplotypes 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 (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 mainland Brazil are mainly made up of turtles from Ascension Island, Trindade Island, Aves Island, and Suriname (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).

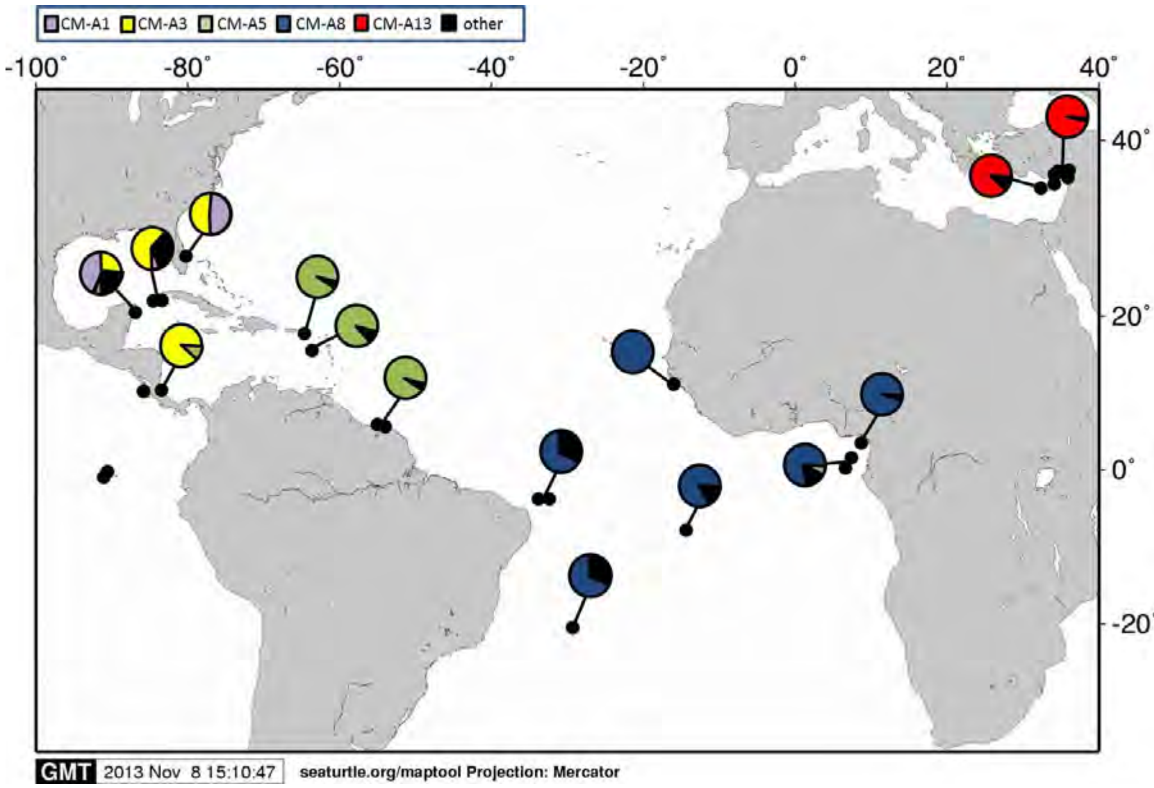

Figure 4.2. Haplotype (mtDNA) frequencies (colors) at nesting sites in the Mediterranean and Atlantic (M. Jensen, NRC, pers. comm., 2013). Note that the dominant haplotypes in eastern Caribbean nesting sites (green) are distinct from those in other regions, but are most closely related to haplotypes 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).

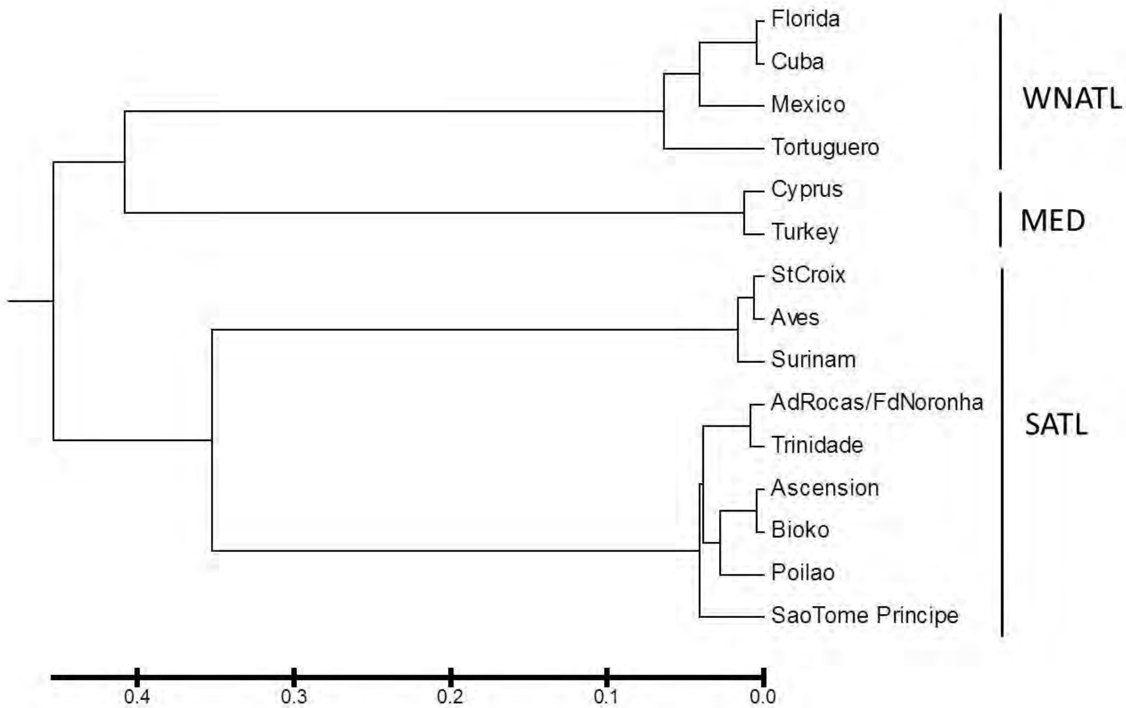

Figure 4.3. Genetic groupings (Neighbor- Joining tree of F_{ST} Values) among green turtle nesting sites in the Mediterranean (MED), western North Atlantic (WNATL) and South Atlantic (SATL). Groupings are based on 396 bp mtDNA sequence data (M. Jensen, NRC, pers. comm., 2013).

In the southwest Indian Ocean, Bourjea *et al.* (2007b) 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).

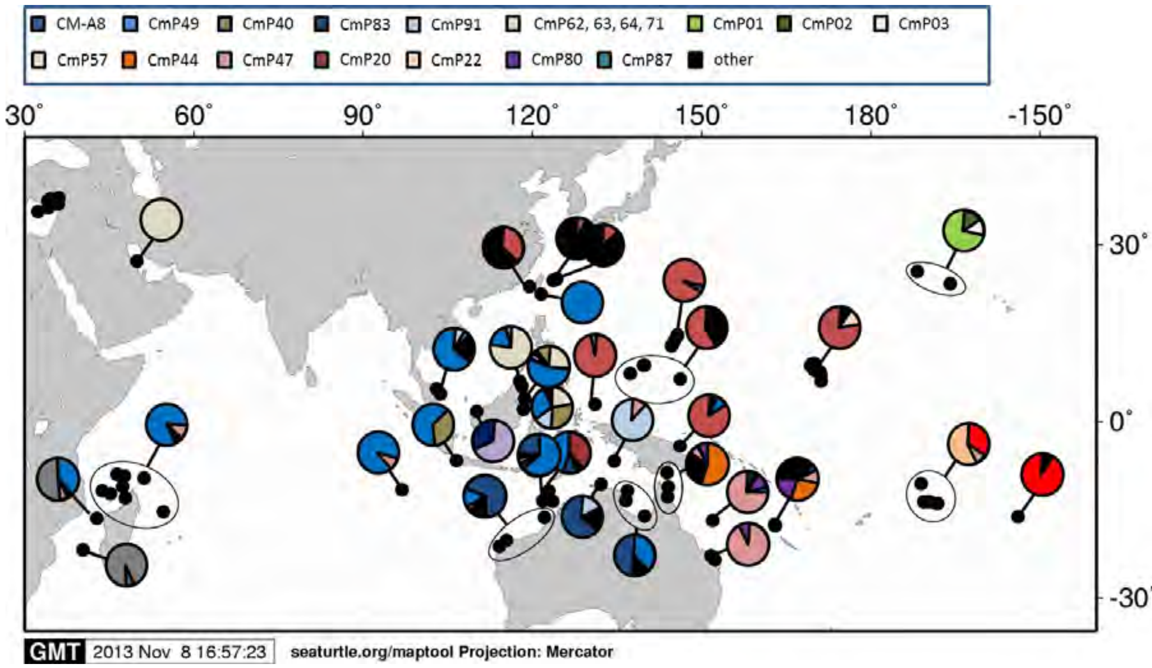

Figure 4.4. Haplotype (mtDNA) frequencies (colors) at sampled nesting sites in the Indian Ocean (M. Jensen, NRC, pers. comm., 2013).

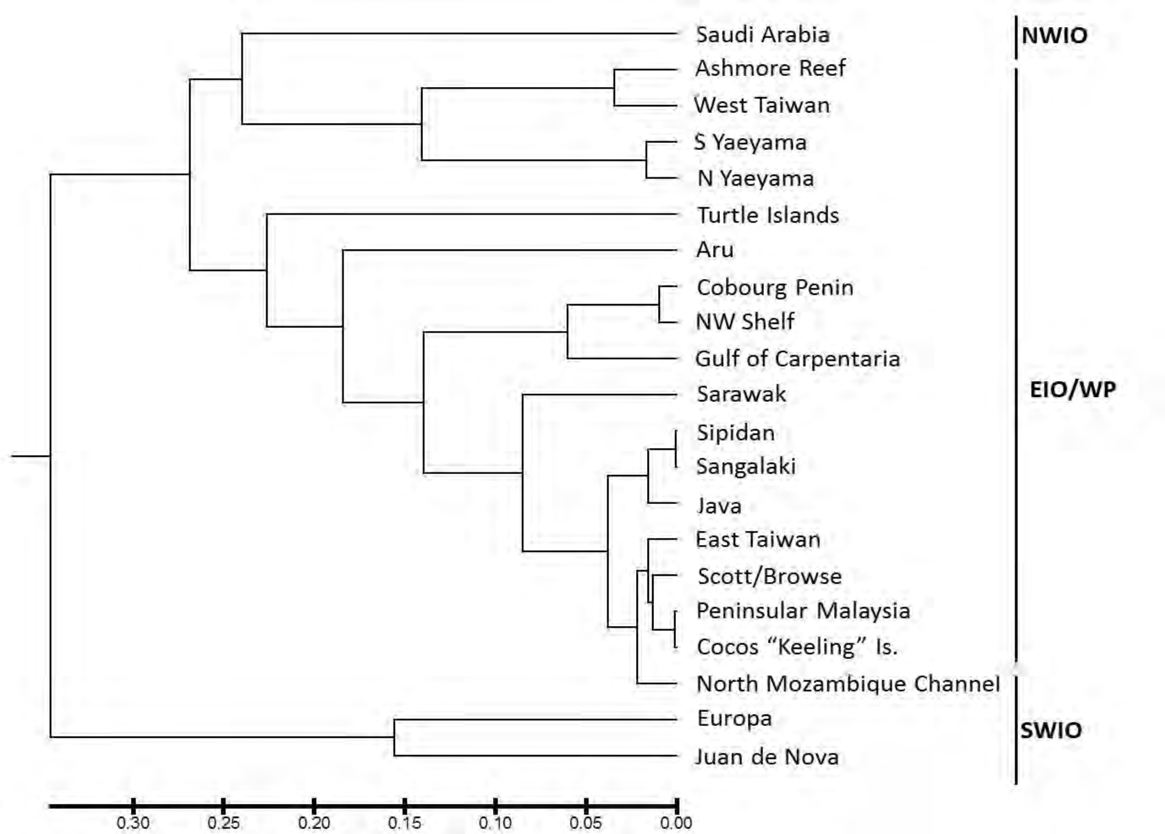

Figure 4.5. Genetic groupings (Neighbor- Joining tree of F_{ST} Values) among green turtle nesting sites in the Indo-Pacific. The Indo-Pacific includes the southwest Indian Ocean (SWIO), northwest Indian Ocean (NWIO) and east Indian and western Pacific Oceans (EIO/WP). Groupings are based on 396 bp mtDNA sequence data (M. Jensen, NRC, pers. comm., 2013).

Bourjea *et al.* (2007b) 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 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.

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), 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 shows 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, 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).

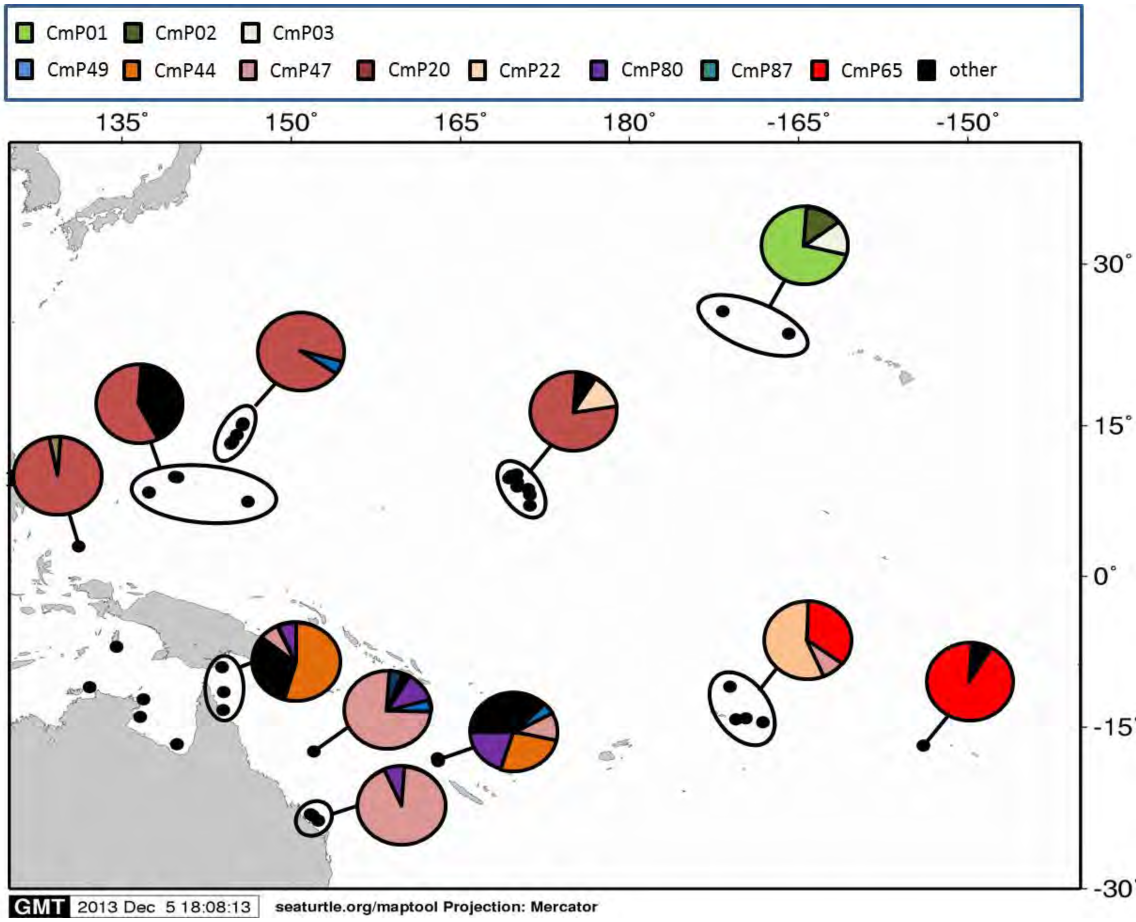

Figure 4.6. Haplotype (mtDNA) frequencies (colors) at sampled nesting sites in the western Pacific (M. Jensen, NRC, pers. comm., 2013).

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 GBR 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 Central West 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_{ST} 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; Dutton *et al.*, 2014; M. Jensen, NRC, pers. comm., 2013; Figure 4.7). Nesting sites in Central West Pacific show very limited connectivity with surrounding regions (Figure 4.6).

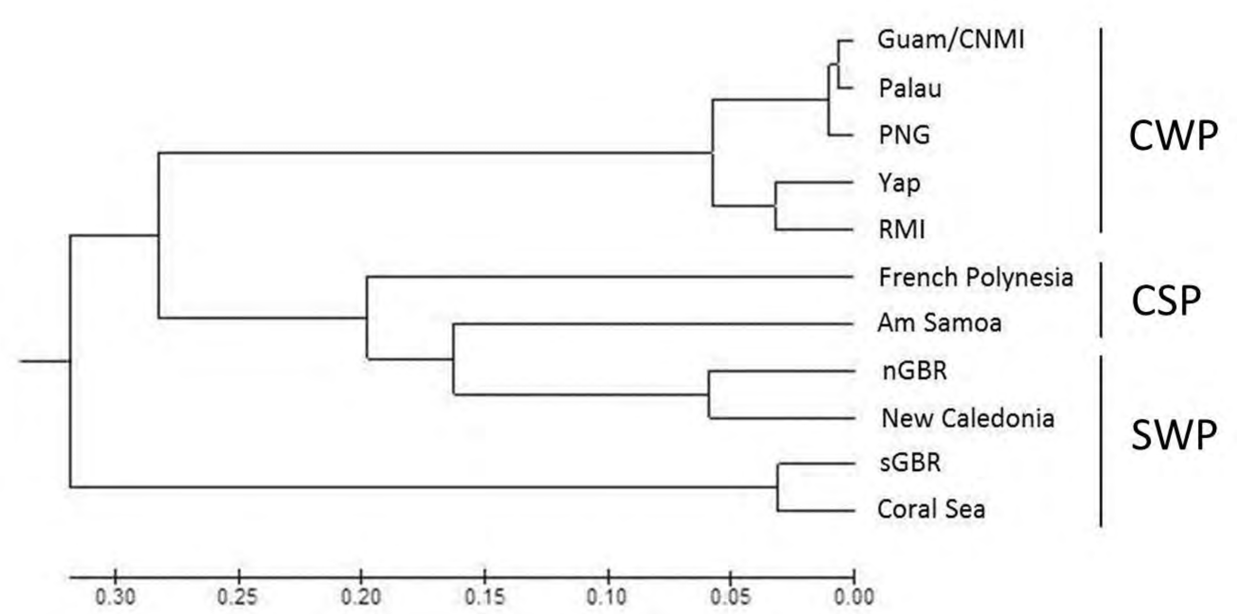

Figure 4.7. Genetic groupings (Neighbor-Joining tree of FST Values) among green turtle nesting sites in the western Pacific. Western Pacific includes the central west Pacific (CWP), central South Pacific (CSP), and southwest Pacific (SWP). Relationships based on 384 bp of control region (M. Jensen, NRC, 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 hatchling 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 Galapagos Islands.

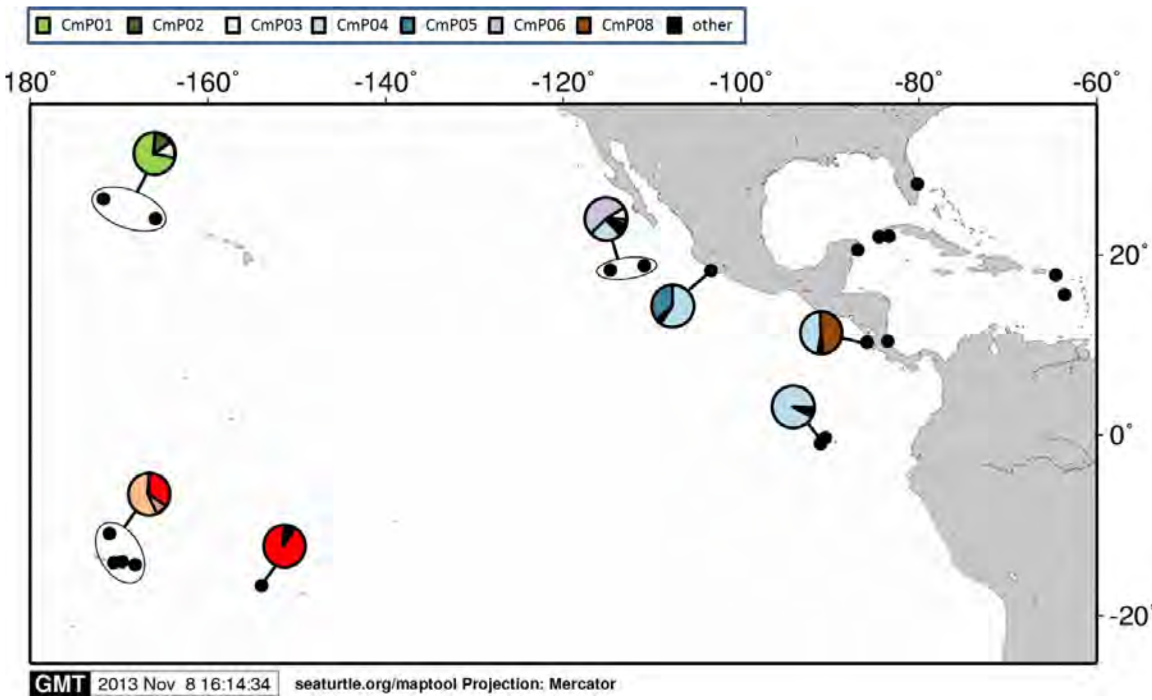

Figure 4.8. Haplotype (mtDNA) frequencies (colors) at sampled nesting sites in the Pacific (P. Dutton, NMFS, pers. comm., 2013; M. Jensen, NRC, pers. comm., 2013).

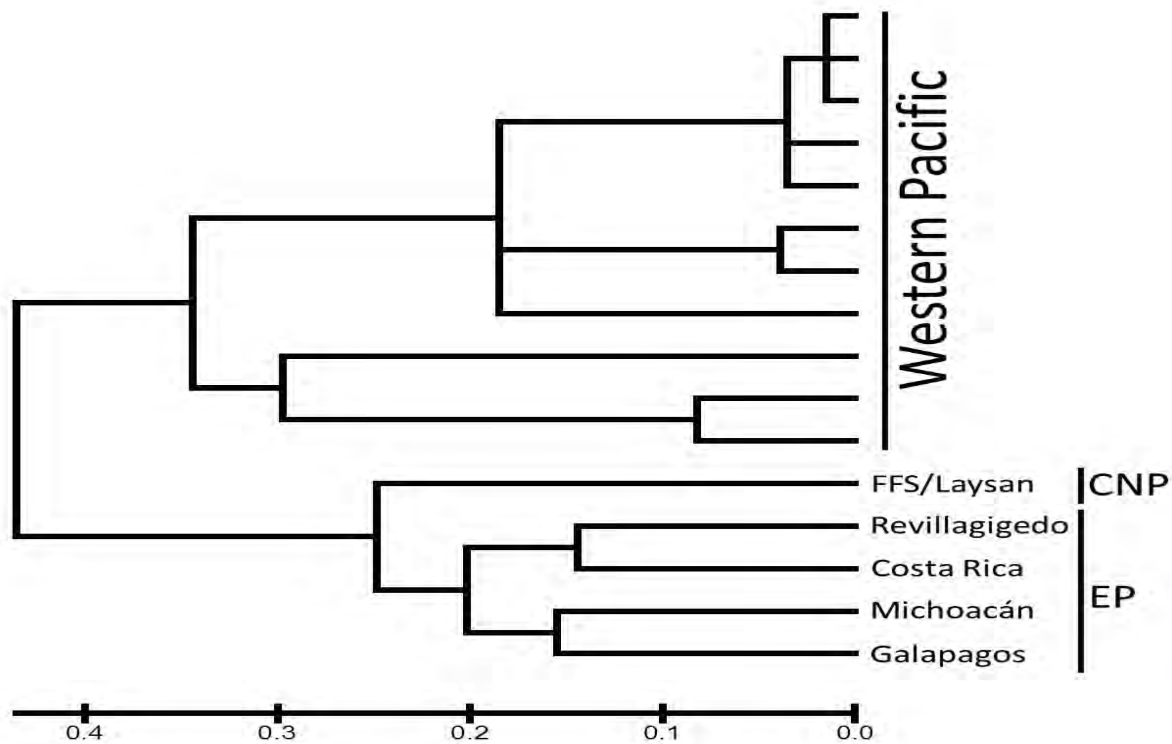

Figure 4.9. Phylogenetic groupings (FST Values) among central North Pacific (CNP) and eastern Pacific (EP) green turtle nesting sites. Relationships based on 384 bp of control region relative to those in the western Pacific. (M. Jensen, NRC, pers. comm., 2013).

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 suggests 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, 2003; 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 insights that are 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 of the 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 the South Atlantic regions and no mixing with the Mediterranean region. 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 (Troëng *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 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, Republic of Honduras (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, Honduras, Belize, and Mexico (Troëng *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).

4.1.1.2.1.2. 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, 2012).

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.

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), Libya, and Turkey—with movements largely restricted to the eastern Mediterranean (Godley *et al.*, 2002b; 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.*, 2002b; 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, Oman, and in 1999 on the Dimaniyat Islands, Oman 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 green turtle movements (Ross and Barwani, 1982; Ross, 1987; Salm, 1991). Some turtles in the area migrate long distances from distant feeding grounds to nesting beaches, while others are quite sedentary. Tagging studies have revealed that some turtles nesting on Ras al Hadd and Masirah can be found as far away as Somalia, Ethiopia, Yemen, Saudi Arabia, and the upper Gulf, and Pakistan (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.

4.1.1.2.1.5. Southwest Indian Ocean

Tagging

Evidence from tag returns indicates that 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 areas 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 internesting 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 genetics, range and movements, as evidenced by genetic, 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 hatchling 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.*, 2012), 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 Enu, 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^{o} 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/SP, 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). In the norther hemisphere, juvenile turtles can be found as far north as Cape Cod Bay, Massachusetts, as Bermuda to the east, and throughout the Caribbean. Water temperatures below 8°C result in hypothermic cold-stunning (Witherington and Ehrhart, 1989b) 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 Ehrhart, 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.

Pacific Ocean

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 CO_2 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 GBR, Japan, Coral Sea, and through Southeast Asia in low frequency; Clade V (purple) found only in nesting sites in the Coral Sea (GBR, 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 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 barriers 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.

A summary of information used to determine discreteness is depicted in Table 4.1, below.

Table 4.1. Summary of the spatial separation, demography, tagging and genetics used to determine discreteness.

DISCRETENESS				
DPS	**Spatial Separation (*physical*)**	**Demography *(physiological)***	**Tagging *(behavioral)***	**Genetics**
1. North Atlantic	Some overlap at southern edge of N Atl range w/ DPS 3; no overlap with DPS 2		Minimal transboundary recoveries (some w/ DPS 3, no transboundary tag recoveries w/ DPS 2); localized movements; distinct FP phylogeny compared to DPS 3	N Atl haplotypes found juveniles captured in Brazil and Argentina (DPS 3); no genetic structure from nDNA w/ DPS 3, but a small number of genetic markers were examined
2. Mediterranean	Only population in entire sea basin	Second smallest MNS of any region (after EP)	No transboundary recoveries w/ DPS 1; localized movements; no immigration from DPS 1 despite extensive data	Clear genetic differences w/ DPS 1

DISCRETENESS				
DPS	**Spatial Separation (*physical*)**	**Demography *(physiological)***	**Tagging *(behavioral)***	**Genetics**
3. South Atlantic	Some overlap at northern edge of range w/ DPS 1	Largest MNS globally	Extensive movements within region, but no immigration or emigration revealed through satellite telemetry w/ DPS 1 or 4; distinct FP phylogeny compared to DPS 1	Haplotype frequencies provide no evidence for contemporary connectivity around Cape of Good Hope; haplotypes from turtles in Mozambique Channel are from the same clade as those in S. Atlantic, but this reflects distant evolutionary history, high local connectivity
4. SW Indian	Cape of Good Hope separates from DPS 3; no clear current boundaries w/ DPS 5 or 6; apparent nesting gap w/ DPS 3, 5, 6	MNS larger than DPS 5 or 6	Minimal transboundary recoveries w/ DPS 5; no transboundary recoveries w/ DPS 3; no transboundary recoveries and minimal data w/ DPS 6; localized movements despite extensive data; no immigrations but minimal data (DPS 6)	Genetic differences present but not strong (DPS 6), no nDNA, just mtDNA (DPS 6), Strong genetic differences (DPS 3,5)
5. N Indian	Apparent nesting gap w/ DPS 4, 6	MNS smaller than DPS 4, 6	Minimal transboundary recoveries w/ DPS 4; no transboundary recoveries w/ DPS 6; localized movements	Globally unique clade; clear genetic differences w/ DPS 4, 6; almost all rookeries in N Indian un-sampled

DISCRETENESS				
DPS	**Spatial Separation (*physical*)**	**Demography *(physiological)***	**Tagging *(behavioral)***	**Genetics**
6. E Indian- W Pacific	Wallace Line is biogeographic boundary w/ DPS 7); apparent nesting gap w/ DPS 5; large distance from DPS 4; oceanographic currents suggest possible connectivity w/ DPS 4	MNS larger than DPS 5	Moderate transboundary recoveries w/ DPS 7, 8; rare transboundary recoveries w/ DPS 9; localized movements; no transboundary recoveries w/ DPS 10	Globally unique haplotypes; distinct and high nucleotide diversity; clear genetic differences from DPS 5, 7-9; historical genetic connectivity w/ DPS 4
7. CW Pacific	Wallace Line - biogeographic boundary w/ DPS 6; oceanographic boundary w/ DPS 10		Moderate transboundary movements, although small sample size; moderate transboundary recoveries w/ DPS 6; minimal transboundary recoveries w/ DPS 8	Globally unique haplotypes, clear genetic differences w/ DPS 6, 8, 9; AMOVA supports stand-alone entity; no genetic immigration from DPS 8 or 9
8. SW Pacific	Closely proximate DPSs		Moderate transboundary recoveries w/ DPS 6; minimal transboundary recoveries w/ DPS 7, 9; localized movement, although small sample size	Globally unique haplotypes; oldest haplotype lineages; distinct and high nucleotide diversity; clear genetic differences w/ DPS 6, 7, 9; nDNA and mtDNA distinctiveness w/ DPS 6, 7, 9

DISCRETENESS				
DPS	**Spatial Separation (*physical*)**	**Demography *(physiological)***	**Tagging *(behavioral)***	**Genetics**
9. CS Pacific	Oceanographic barrier w/ DPS 10, 11; EP turtles found in Am Samoa longline (DPS 11)	Data deficient	Localized movements although limited data; modest transboundary recoveries w/ DPS 7, 8; minimal tranboundary immigration from DPS 7, 8; no recoveries w/ DPS 10	Clear genetic differences w/ DPS 7, 8, 10, but only two rookeries sampled
10. CN Pacific	Most isolated archipelago globally; oceanographic barrier w/ DPS 11; large distances to DPS 7-9, 11	MNS larger than DPS 11	Rare transboundary recoveries but extensive data w/ DPS 6-9, 11; localized movements w/ extensive data	No shared haplotypes w/ DPS 6-9; shared haplotype with DPS 11
11. E. Pacific	Moderate numbers of juveniles found in DPS 7, 8 and high seas of CNP (DPS 10); CSP (DPS 9) 'yellow' juveniles found in southeastern EP	Smallest MNS of any DPS; mostly black in color	No tag recoveries or satellite tracks of EP turtles outside EP, although small number of EP turtles found in DPS 7, 8 and10; no tag recoveries or satellite tracks of turtles from other DPSs in EP, although small number of turtles from DPS 9 found in EP	Clear genetic differences w/ DPS 7-10; some shared haplotypes w/ DPS 10

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 4.2) showed that each of these population units received at least 70 percent 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 4.1 depicts these units. The next section explains how each of these population units was evaluated in terms of significance.

Table 4.2. Results of SRT voting for discreteness. Values show the mean of affirmative likelihood points (with range among SRT members in parentheses).

	DPS	Discreteness
1	North Atlantic	83.8 (55-95)
2	Mediterranean	96.5 (90-100)
3	South Atlantic	84.1 (60-90)
4	Southwest Indian	72.9 (50-90)
5	North Indian	81.3 (65-95)
6	East Indian-West Pacific	71.2 (40-90)
7	Central West Pacific	70.9 (50-90)
8	Southwest Pacific	79.9 (40-99)
9	Central South Pacific	70.0 (50-95)
10	Central North Pacific	93.7 (85-100)
11	East Pacific	91.6 (75-100)

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 central North and central South Pacific and in the southern Indian Ocean present challenges for turtles foraging in those areas.

In the central South 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 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. With regard to the latter, the loss of turtles from this large and complex area, which 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

Neutral genetic markers were already used heavily in defining discreteness, however, genetic uniqueness is also germane to the significance of numerous discrete units. Numerous discrete units have globally unique haplotypes, indicating that loss of these units would be a significant genetic loss to the species as a whole: North Atlantic, Mediterranean, South Atlantic, North Indian, Central West Pacific, Central South Pacific, Central North Pacific and East Pacific. Two discrete units have ancestral haplotypes, also making their potential loss a significant loss to the species: East Indian-West Pacific and Central West Pacific.

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 Mediterreanean. For example, in a comparison of average nester sizes Hirth (1997) determined that nesters at Isla Trindade, Brazil (average CCL 115.2 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 (1997), 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). 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).

Behavior

Because of the temperate nature of many green turtle foraging areas at the northern and southern extents of their range in the East Pacific, green turtles may experience colder waters in this region than anyplace 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 behavioural 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.

A summary of information used to determine significance is depicted in Table 4.2, below.

Table 4.3. Summary of the ecological setting, gap in range, and marked genetics used to determine significance.

SIGNIFICANCE				
DPS	**Ecological Setting**	**Gap in Range**	**Marked Genetics**	**Other**
1. North Atlantic	Caribbean sea unique w/ expansive seagrass beds, broad continental shelf; Nesting in N. FL to NC outside normal latitudinal range	No gene flow w/ Med (DPS 2); some gene flow with DPS 3	Distinct genetic differences based on mtDNA (DPS 2,3); some globally unique haplotypes	2-year remigration interval; high incidence of FP
2. Mediterranean	Unique habitat—enclosed sea, low-productivity waters, most saline waters, northern-most nesting	Encompasses large region; apparent biogeographic boundary of W. Med would hinder re-population	100% globally unique haplotypes; significant difference in mtDNA markers from DPS 1	Second smallest MNS of any region (after EP); northern-most latitude for nesting
3. South Atlantic	Ascension Isl. is only mid-ocean ridge nesting site	Population encompasses vast region (S. hemisphere of ocean basin)	Globally unique haplotypes	Largest MNS globally
4. SW Indian	Major cold water upwelling in the Mozambique Channel creates distinctive habitat	No known immigration from DPS 3, 5, 6; apparent biogeographic barrier w/ DPS 3		Largest MNS for Indian Ocean
5. N Indian	Unique habitat w/ heat adapted coral in Persian and Red Seas; high saline waters	Isolated and far from adjacent DPSs (4 and 6)	Limited genetic data from one nesting population shows globally unique and very divergent haplotypes in Saudi Arabia	
6. E Indian- W Pacific	Most extensive continental shelf globally; high rainfall and extensive river runoff produce low salinity water in the N Indian Ocean	Population encompasses large region; loss would create major connectivity gap between DPSs 4-5 and 7-8	Ancestral haplotypes; significant mtDNA diversity	

SIGNIFICANCE				
DPS	**Ecological Setting**	**Gap in Range**	**Marked Genetics**	**Other**
7. CW Pacific		Apparent oceanic boundary w/ DPS 10; apparent biogeographic boundary w/ DPS 6	Globally unique haplotypes	
8. SW Pacific	GBR provides unique habitat; periodic isolation over geological time		Ancient lineage; significant mtDNA diversity	
9. CS Pacific	Nesting on small atolls and islands and more spread out than elsewhere (no nesting stronghold)	Population encompasses large oceanic region; apparent oceanic boundary w/ DPS 10	A single, globally unique haplotype; extensive sampling in other regions has not detected haplotype	
10. CN Pacific	No continental shelf, only mid-basin oceanic pinnacles	Encompasses large oceanic region; most isolated of all DPSs; apparent biogeographic boundary w/ DPS 11 and oceanic boundary w/ DPS 7, 9	Globally unique haplotypes; extensive sampling in other regions has not detected haplotypes; historic gene flow w/ DPS 11	High incidence of FP; basking
11. E. Pacific	Unique diet due to very narrow continental shelf and low levels of seagrass; equatorial upwelling (ENSO)	Very large range; apparent biogeographic boundary w/ DPS 10	Globally unique haplotypes; extensive sampling in other regions has not detected haplotypes; historic gene flow w/ DPS 10	Smallest MNS of all regions; unique overwintering behavior; basking in Galapagos

After considering all of the above information, the SRT voted on significance, and each discrete population unit identified earlier received a substantial majority (65 percent or more) affirmative votes (Table 4.4). 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 4.4. Results of SRT voting for significance. Values show the mean of affirmative likelihood points (with range among SRT members in parentheses).

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

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.

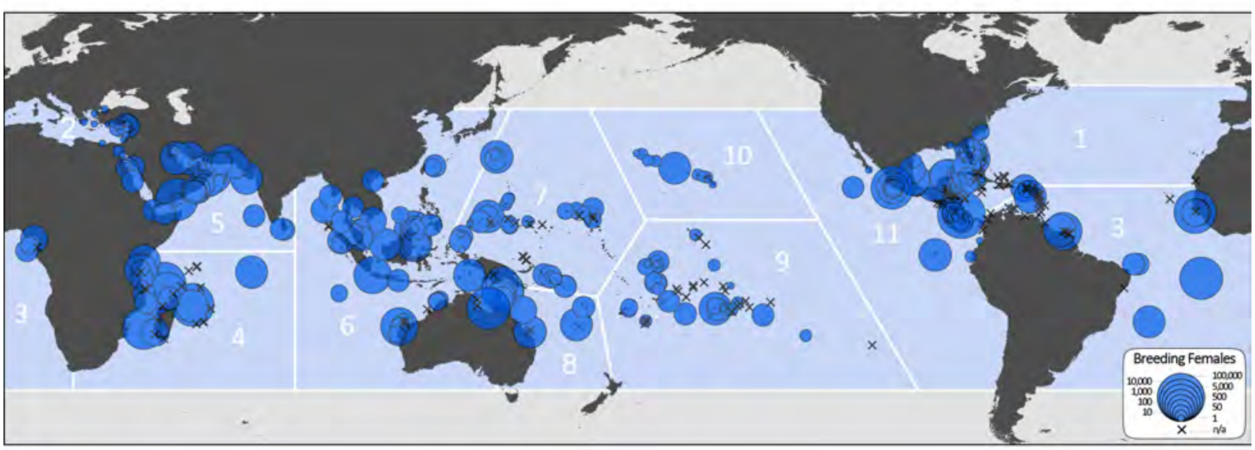

Figure 4.10. Map of all C.mydas nesting sites indicating delineation of DPSs.

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 water 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 most saline water 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 matrilines 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 (Clade I), (2) populations from the Gulf of Mexico/Central America have a low frequency of haplotypes from the South Atlantic Clade (Clade II), and (3) 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) but show 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

This region is unique in that it 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. The isolation and distance from other Indian discrete populations would render its loss a significant gap in the species. Nesting turtles here are smaller than in other Indian DPSs, indicating possible genetic adaptions to local environmental conditions.

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 4.4). 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.

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 Tables 4.1 and 4.2 above).

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 gap 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 percent) 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 (7.5°N, 77°W), north to 10.5°N, 77°W, then extending due east across the Atlantic Ocean at 19°N latitude to the African continent, and extending north along the western coasts of Africa and Europe (west of 5.5°W longitude) to 48°N latitude (Figure 5.1).

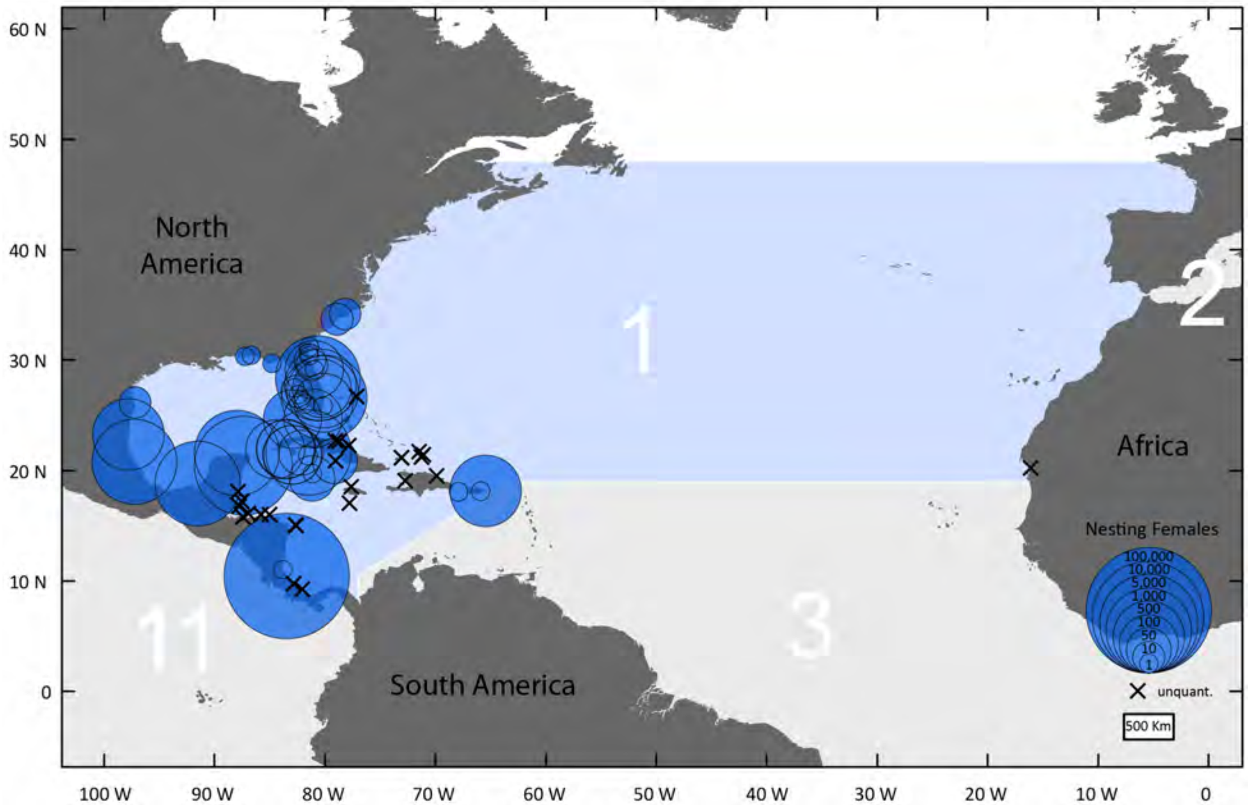

Figure 5.1. Geographic range of the North Atlantic DPS. Size of circles indicates estimated nester abundance (see Section 5.2.1). Locations marked with '×' indicate nesting sites lacking abundance information.

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, Canarreos 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).

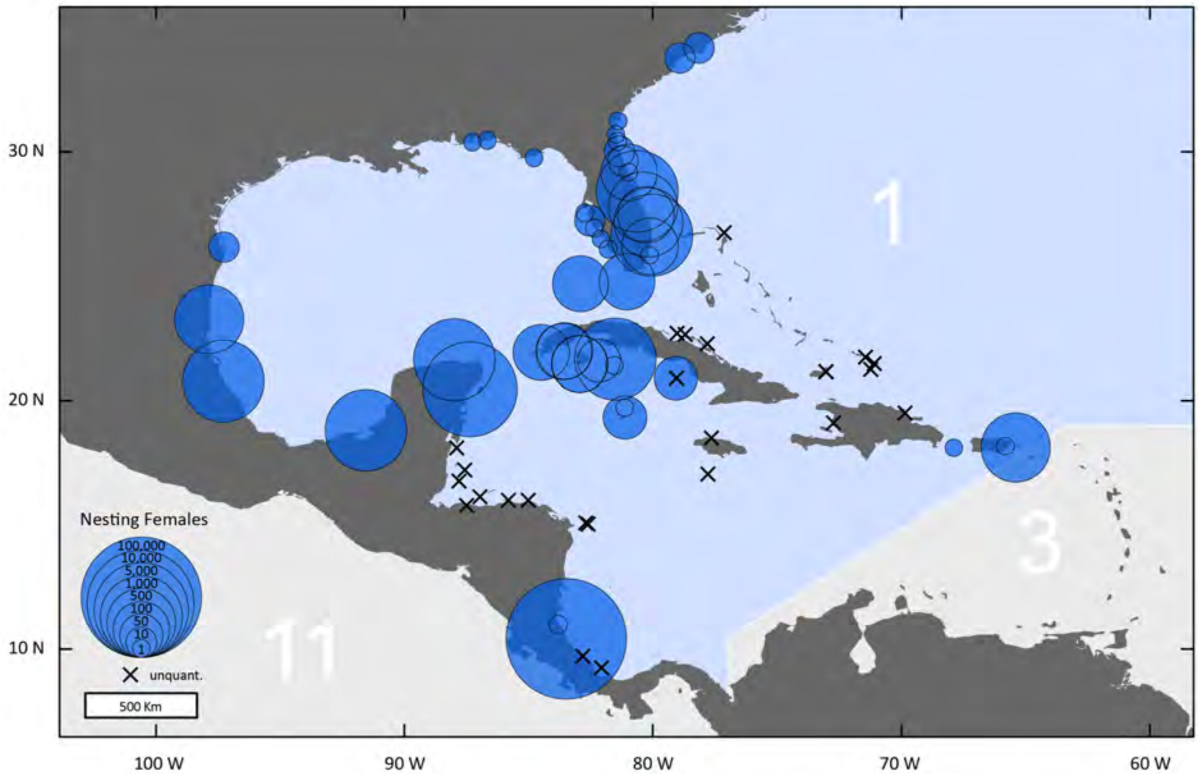

Figure 5.2. Close up of nesting distribution of green turtles in the western North Atlantic DPS. Size of circles indicates estimated nester abundance (see Section 5.2.1). Locations marked with '×' indicate nesting sites lacking abundance information.

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, 2002; Schroeder *et al.*, 1998; Ehrhart *et al.*, 2007) to man-made embayments (Redfoot and Ehrhart, 2000; Kubis *et al.*, 2009). Turtles feed primarily on seagrass 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.*, 2013), and they change habitats during successive stages of life (Bagley *et al.*, 2008; Reich *et al.*, 2008; Vander Zanden *et al.*, 2013). 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 (Groombride 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.*, 2003a).

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 73 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: Adult Female Abundance = (nests/clutch frequency) * 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 [(total counted females / years of monitoring) x 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.

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Cayman Islands	Grand Cayman	2005-2009	72
Cayman Islands	Little Cayman	2007	5
Costa Rica	Tortuguero	2009-2011	131,751
Cuba	Cayo Largo (Eastern Keys of Isla de la Juventud)	2008-2010	1,284
Cuba	Beaches of the Guahanacabibes Peninsula	2010-2012	201
Cuba	South Isla de la Juventud	2010-2011	170
Cuba	San Felipe	2009-2011	162
Cuba	Guanal	2009-2011	124
Cuba	Cayo Siju, Cayo Real, Juan Garcia (Cayos de San Felipe)	2007-2009	123
Cuba	Playas Archipiélago Jardines de la Reina	2011	88
Cuba	Eastern Keys of Isla de la Juventud	2010	64
Cuba	Cayo Rosario	2008	10
Mexico	Quintana Roo	2010-2012	18,257
Mexico	Campeche (includes Isla Aguada; see Figure 5.5.)	2010-2012	2,207
Mexico	Yucatan	2006-2011	2,111
Mexico	Veracruz	1998-2000	1,040
Mexico	Tamaulipas	2009-2010	715
Nicaragua	El Cocal	2000	6
Puerto Rico*	Vieques	2010-2012	626
Puerto Rico*	Mona Island	2012	12
Puerto Rico*	Humacao	2012	6
USA, FL	Brevard County	2011-2012	3,979
USA, FL	Palm Beach County	2011-2012	2,006
USA, FL	Martin County	2011-2012	998
USA, FL	Indian River County	2011-2012	504

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
USA, FL	St. Lucie County	2011-2012	233
USA, FL	Volusia County	2011-2012	215
USA, FL	Broward County	2011-2012	157
USA, FL	Monroe County	2011-2012	120
USA, FL	Flagler County	2011-2012	39
USA, NC	North Carolina	2011-2012	39
USA, FL	Sarasota County	2011-2012	21
USA, FL	St. Johns County	2011-2012	20
USA, TX	Texas	2011-2012	16
USA, SC	South Carolina	2011-2012	11
USA, FL	Lee County	2011-2012	9
USA, GA	Georgia	2011-2012	5
USA, FL	Miami-Dade County	2011-2012	5
USA, FL	Charlotte County	2011–2012	3
USA, FL	Escambia County	2011–2012	3
USA, FL	Collier County	2011–2012	2
USA, FL	Nassau County	2011–2012	2
USA, FL	Okaloosa County	2011–2012	2
USA, FL	Duval County	2011–2012	1
USA, FL	Franklin County	2011–2012	1
USA, FL	Manatee County	2011–2012	1
USA, FL	Walton County	2011–2012	1

* These sites were added to the table following the votes on the critical assessment elements and the probability of reaching quasi-extinction, 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 didn'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.

NESTER ABUNDANCE	# NESTING SITES DPS 1
Unquantified*	26
1–10	16
11–50	7
51–100	3
101–500	9
501–1000	4
1001–5000	6
5001–10000	0
10001–100000	1
>100,000	1
Total Sites	**73**
Total Abundance	**167,424**
PERCENTAGE at Largest NESTING SITE	**79% (Tortuguero, Costa Rica)**

* Not included in Table 5.1

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.

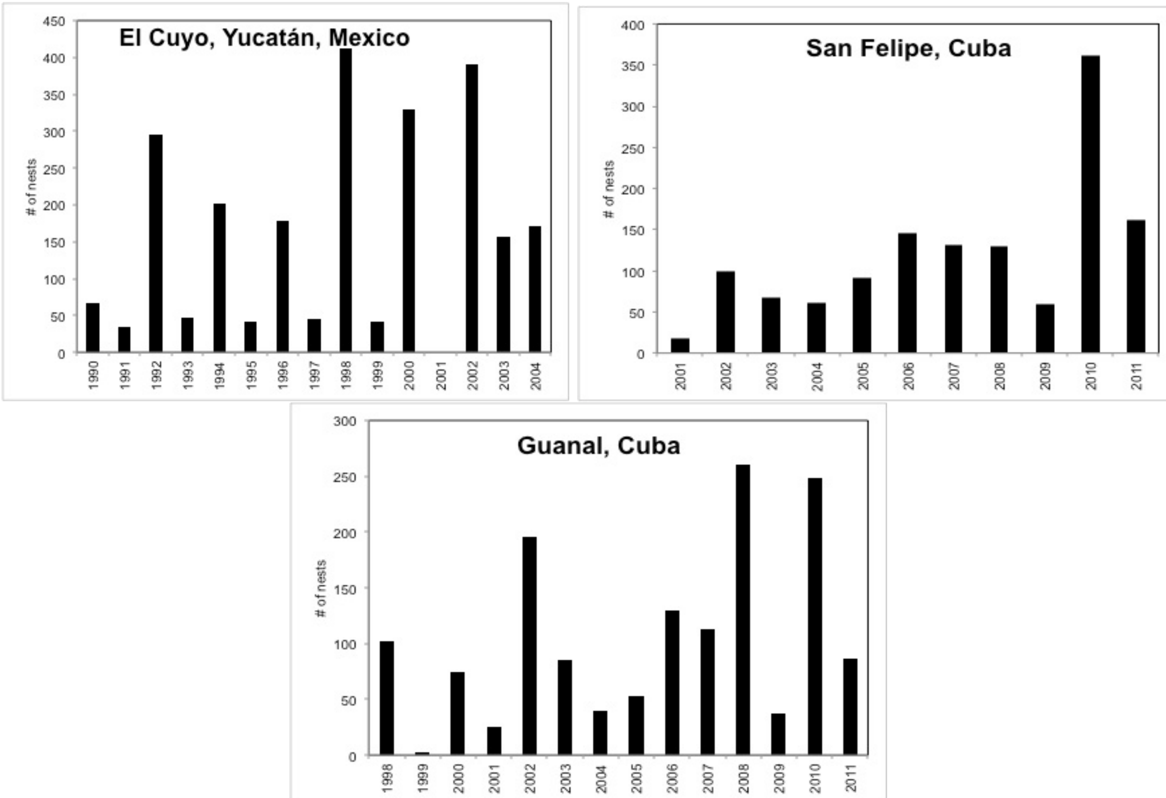

Figure 5.3. Trend data for green turtle nesting in the North Atlantic DPS with greater than 10 yrs of recent monitoring data, with a missing year. These include El Cuyo, Mexico (14 yrs), San Felipe, Cuba (11 yrs), and Guanal, Cuba (14 yrs).

Of the three sites with bar plots, there were apparent patterns of high-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 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 Lagueux, 2005; Troëng and Rankin, 2005).

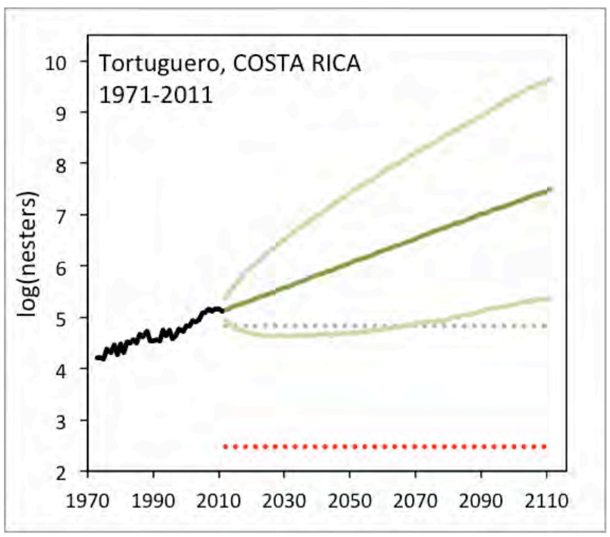

Figure 5.4. Stochastic Exponential Growth (SEG) Model Output at Tortuguero, Costa Rica. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference. 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.

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).

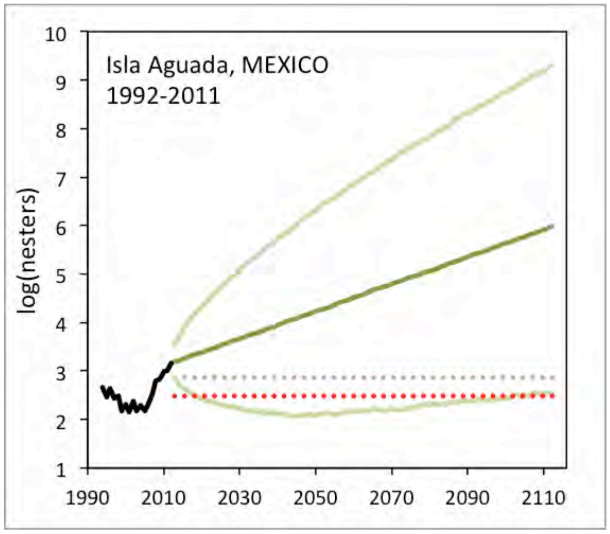

Figure 5.5. Stochastic Exponential Growth (SEG) Model Output for Isla Aguada, Mexico. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference. Number of nesters was computed from nests using 4.01 nests per female (Guzmán-Hernández and García Alvarado, 2013b).

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 (2013b) (Figure 5.6). The units for the Guanahacabibes indices are expressed as adult females, so no transformation from nests to nesters was needed.

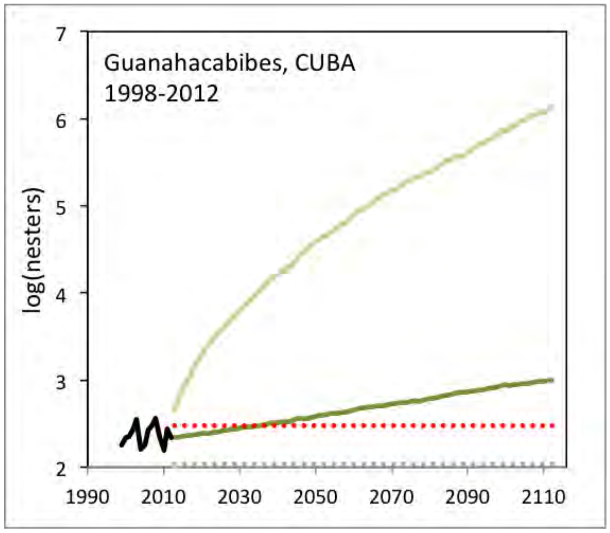

Figure 5.6. Stochastic Exponential Growth (SEG) Model Output for Guanahacabibes, Cuba. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference.

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.

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 (Florida Fish and Wildlife Conservation Commission, 2012; Figure 5.7).

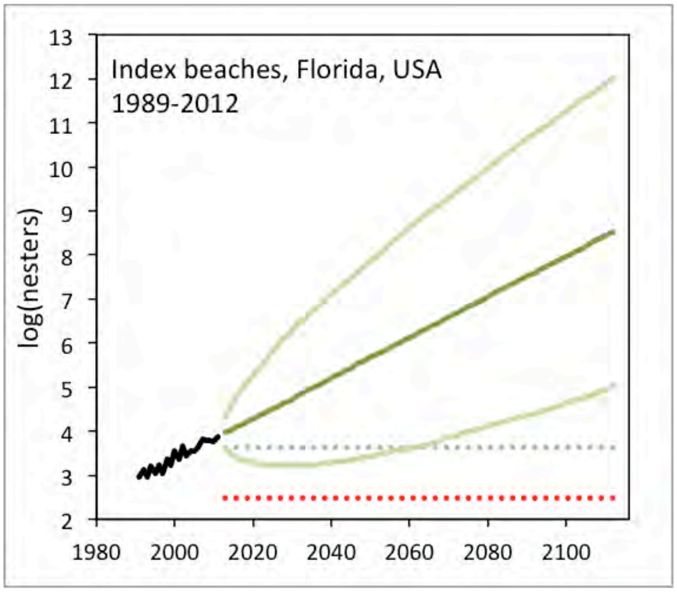

Figure 5.7. Stochastic Exponential Growth (SEG) Model Output for Florida Index Beaches. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference. 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 increases 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 Península and Cayería 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. In addition, studies using nuclear DNA markers raised the possibility of connectivity via male-mediated gene flow among Atlantic nesting populations (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 (Troëng *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; Patrício *et al.*, 2011), and Texas (Shaver, 1994, 2002). Nesters have been satellite tracked from Florida, Cuba, Cayman Islands, Mexico, and Costa Rica. 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 within the DPS (Troëng *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 occur (Eaton *et al.*, 2008). These research projects include tracking post-hatchling to adult green turtles. Post-hatchlings and juvenile, 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; Makowski *et al.*, 2006; Ehrhart *et al.*, 2007) and St. Joseph Bay in the Florida panhandle. 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 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 101.7cm 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, 1989b; 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; Carr *et al.*, 1978; Johnson and Ehrhart, 1996; Guzmán-Hernández and García Alvarado, 2009, 2010, 2011, 2013a, 2013b). 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, 1989b), and have a hatching success ranging from 61.6 percent in Florida (Witherington and Ehrhart, 1989b) to 92 percent in Mexico (Xavier *et al.*, 2006), although the high of 92 percent is an overestimate 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 península 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.

All of these factors were considered given current management regimes. 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 Aquada 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.*, 2002). 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 (http://www.dep.state.fl.us/beaches/programs/ccclprog.htm). 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 USFWS, 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 beach and dune sand placement, and sand extraction (Lutcavage *et al.*, 1997; Bouchard *et al.*, 1998; Mosier, 1998; Mosier and Witherington, 2002; Leong *et al.*, 2003; Roberts and Ehrhart, 2007). 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 hatchling 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 2011 sea turtle nesting season in Florida, 2,110 green turtle hatchlings were reported to FWC as being disoriented. In 2010, 17 nesting green turtles were disoriented due to artificial lighting, and 7 nesting green turtles were disoriented in 2011 (R. Trindell, Florida Fish and Wildlife Commission, pers. comm, 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 hatchling sex ratio (Marcus and Maley, 1987; Schmelz and Mezich, 1988). Fallen Australian pines limit access to suitable nest sites and can entrap nesting females (Reardon and Mansfield, 1997). The shallow root network of these pines can interfere with nest construction (Schmelz and Mezich, 1988). 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 Wyneken, 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 which is linked to harmful algal blooms that result 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 be 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 seagrasses, 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 Troëng, 2001), Mexico (Seminoff, 2000; Gardner and Nichols, 2001), Cuba (Fleming, 2001), Nicaragua (Lagueux, 1998; Humber *et al.*, 2014), 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; Fretey, 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 (Troëng, 1998; Troëng 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 (Gonzalez Prieto and Harrison, 2012). In 2012, 32 adult green turtles were documented as being poached on the Tortuguero beach (Gonzalez Prieto and Harrison, 2012). The nesting colony at Tortuguero has exhibited encouraging trends since the early 1990s (Bjorndal *et al.*, 1999; Gonzalez 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. Moncado, 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 number of nests at Tortuguero continues to increase.

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, 2010). 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.* (2004)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 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).

5.2.5.3. 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; Patrício *et al.*, 2011), Cayman Islands (Wood and Wood, 1994; Dow *et al.*, 2007), Costa Rica (Tortuguero; Mangel and Troëng, 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 (Hirama, 2001; Ene *et al.*, 2005; Foley *et al.*, 2005; Hirama and Ehrhart, 2007).

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 Schroeder *et al.*, 1998; Hirama and Ehrhart, 2007). 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 habitat containing potential contaminants, no direct link has been established. Interestingly, however, the disease remains absent at 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 Balazs, 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.*, 2008).

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 raccoons, feral hogsfoxes (*Urocyon cinereoargenteus* and *Vulpes vulpes*), coyotes (*Canis latrans*), armadillos (*Dasypus novemcinctus*), and red fire ants (Stancyk, 1982). At Tortuguero National Park, green turtles killed by jaguars have increased since 1997, with 57 killed in 2011 (Gonzalez 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) found that ants invaded 35 of 237 (14.8 percent) 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.

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.

5.2.5.4. Factor D: Inadequacy of Existing Regulatory Mechanisms

Regulatory mechanisms to protect green turtles are in place to varying degrees throughout this DPS, and include State, Federal, and international laws and mechanisms. However, some regulatory mechanisms 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).

The analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels.

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 USFWS, 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. 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.

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 (Mansfield *et al.*, 2002). However, green 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 to 1997 and 2001 to 2003. 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 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 occurrences 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.*, 2014). 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 hatchling 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.

Along the southeastern U.S., one climate change model predicted 1 meter sea level rise (SLR) by 2060 resulting in the inundation of more than 50 percent of coastal wildlife refuges (Flaxman and Vargas-Moreno, 2011). Considering green turtle nesting in Florida is concentrated along coastal wildlife refuges in southern Florida such as Hobe Sound National Wildlife Refuge and the Archie Carr National Wildlife Refuge, this increase will result in the permanent loss of current green turtle nesting habitat. Loss of beach is expected to be exacerbated by the increase in hurricane frequency and intensity (Flaxman and Vargas-Moreno, 2011).

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 hatchling 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 Ehrhart, 1982; Witherington and Ehrhart, 1989a; 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

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 and state 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. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

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 range from 36 nests/km (22 nests/mi) to 262 nests/km (419 nests/mi) in the Refuge (D. Bagley, University of Central Florida, pers. comm., 2014; K. Kneifl, USFWS, pers. comm., 2014). 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).

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 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 (MDRPRA; 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 (Bjorndal and Bolten, 2010).

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

Law 4/1989 (3/27/1989) on the Protection of the Natural Habitat and the Wild Flora and Fauna; Law 12/1994 from 19 December 1994 on the Natural Areas of the Canaries; Decree 161/97 of the delegations in environmental policy to the Island Councils (Fretey, 2001).

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 Presidental 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

The current legislation protecting sea turtles is Law 307-04 (year 2004), which regulates fisheries and creates the Council for Fisheries and Aquaculture (CODOPESCA). This law prohibits the explotation of all biological aquatic resources, marine or from inland waters as well as those that enjoy legal protection in the Dominican Republic or in any treaty to which the country is signatory or those resolutions CODOPESCA can issue by virtue of this law. Marine mammals, sea turtles and freshwater turtles are included in this category. In addition, in 2012 a new presidential decree (288-12) was issued to protect sea turtles, their eggs and tortoiseshell crafts for a 10 year period.

Guatemala

Ley General de Pesca y Acuicultura Decreto Nº 80 was passed in 2002 (Bräutigam and Eckert, 2006), and 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

Wildlife Law (1995) and Environmental Law (Ley General de Ambiente No. 41 (1998) protect sea turtles (Bräutigam and Eckert, 2006).

United States

Among the laws in the United States that promote the protection and conservation of sea turtles, the most relevant is the ESA. The 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 green turtles. Both on-the-ground conservation actions, as well as provision of financial and other resources, have resulted in significant population growth of green turtles.

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.

In addition to these national laws, there are State laws and local ordinances throughout the southeastern U.S. and Gulf of Mexico that protect sea turtles, with provisions ranging from lighting ordinances to prohibition of direct harvest.

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 Convention on the Law of the Sea
- 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.

	Critical Assessment Elements					
	Element 1	Element 2	Element 3	Element 4	Element 5	Element 6
	Abundance (1 to 5)	Trends / Productivity (1 to 5)	Spatial Structure (1 to 5)	Diversity / Resilience (1 to 5)	Five-Factor Analyses (–2 to 0)	Conservation Efforts (0 to 2)
MEAN RANK	1.18	1.18	1.45	1.36	-0.45	0.82
SEM	0.18	0.18	0.16	0.20	0.21	0.18
RANGE	1–3	1–3	1–2	1–3	(-2)–0	0–2

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 quasi-extinction under current management regimes 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.

	Probability of Reaching Quasi-Extinction					
	<1%	**1-5%**	**6-10%**	**11-20%**	**21-50%**	**>50%**
MEAN ASSIGNED POINTS	87.00	3.00	1.36	4.09	4.09	0.45
SEM	8.82	1.04	0.73	4.09	4.09	0.45
Min	0	0	0	0	0	0
Max	100	10	7	45	45	5

Of the six categories describing the probability that the North Atlantic DPS will reach quasi-extinction 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 not 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 73 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.

The five-factor analysis highlighted the continuing threats to green turtle habitat that affect 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. Nests 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 harmful 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 percent probability of having at least an 11 percent 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.

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. Given the correlation between application of ESA protections and increasing population trends in the southeast U.S. and the conservation dependence of the species, without alternate mechanisms in place to continue conservation efforts and funding in Florida and beyond, some threats could increase and population trends could be affected.

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 (5.5°W longitude).

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 the Lebanese Republic (Lebanon), Israel, Egypt, and the Hellenic Republic (Greece; Kasparek *et al.*, 2001; Rees *et al.*, 2008; Casale and Margaritoulis, 2010; Figure 6.1).

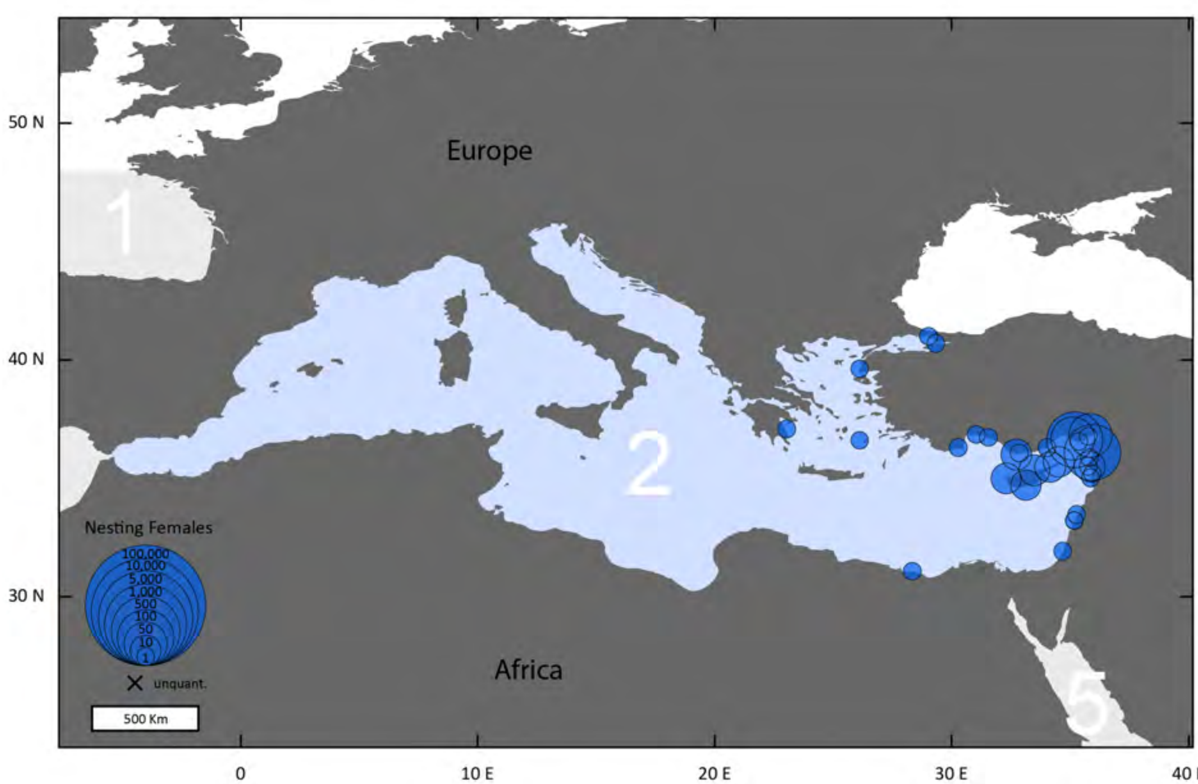

Figure 6.1. Nesting distribution of green turtles in the Mediterranean DPS (water body labeled '2'). Size of circles indicates estimated nester abundance (see Section 6.2.1).

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 Teneketzis, 2003), off Fethiye Beach, Turkey (Türkozan and Durmuş, 2000), along the southeastern coast of Turkey near Syria (Yalçin-Özdilek 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 of Tunisia (Broderick *et al.*, 2007). 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 three nesting concentrations in the Mediterranean from which data are available, including those in Turkey, Cyprus, and Syria. Currently, approximately 452-2,051 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 (*Caretta caretta*)) 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 [(total counted females / year of monitoring) x 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.

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

[2] The region of Cyprus under Turkish control
[3] The region of Cyprus under Greek control

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Turkey	Kizilot	1990	1
Turkey	Anamur	2006	1
Turkey	Goksu Delta	1991-2006	1-7
Turkey	Tuzla	2006	3
Turkey	Karatas	1989	1
Turkey	Agyatan	2006	1
Turkey	Yelkoma	1996	1
Turkey	Yumurtalik	2006	1

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

NESTER ABUNDANCE	# NESTING SITES DPS 2
unquantified	0
1-10	21
11-50	5
51-100	3
101-500	3
501-1000	0
1001-5000	0
5001-10000	0
10001-100000	0
>100,000	0
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). 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 (Figure 6.2). 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.

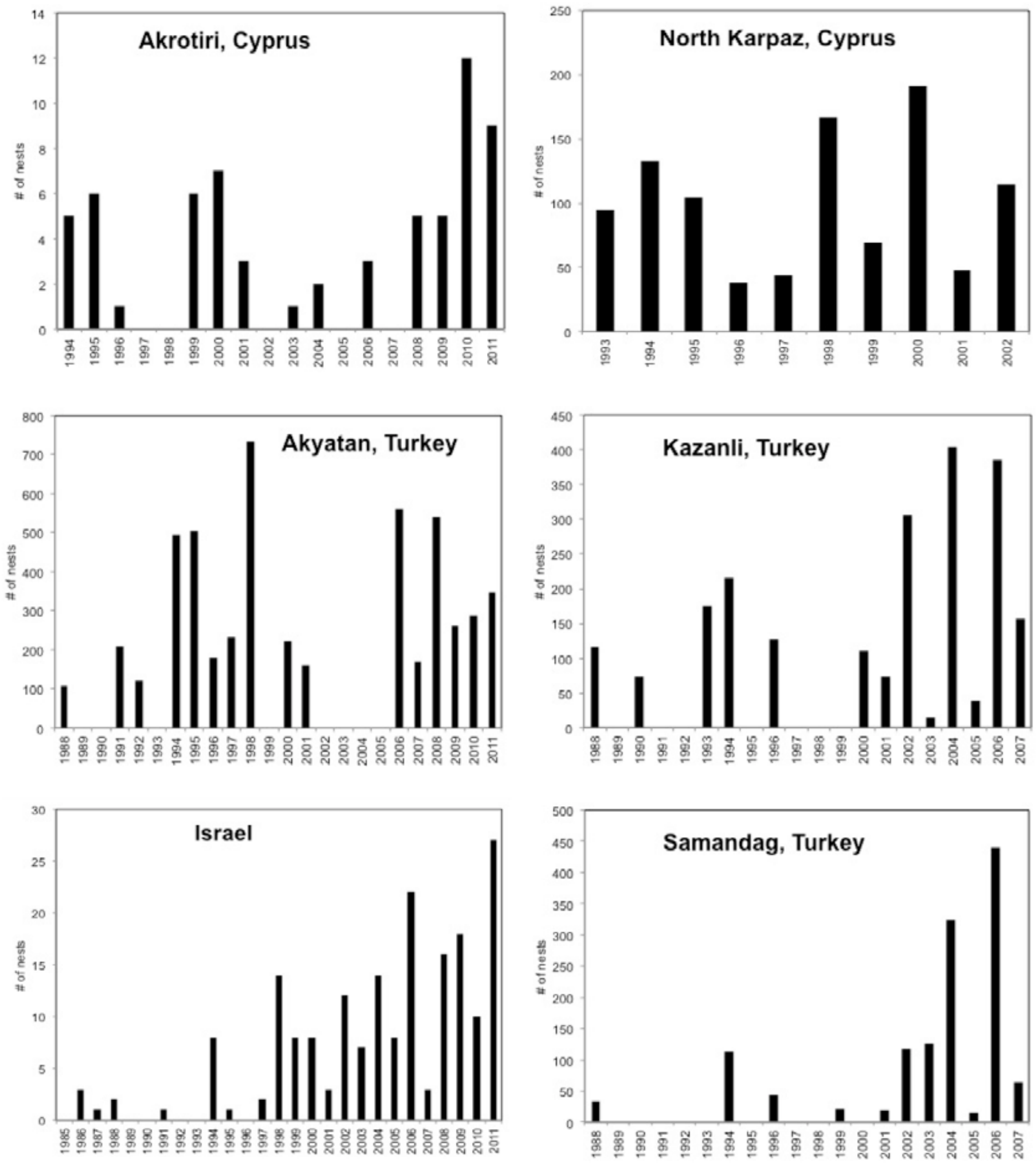

Figure 6.2. Nesting data for green turtle sites in the Mediterranean DPS with greater than 10 yrs of recent monitoring data, although with some missed years. These include Akrotiri, Cyprus (17 yrs), North Carpaz, Cyprus (10 yrs), Akyatan, Turkey (17 yrs), Kazanli, Turkey (13 yrs), Israel (31 yrs), and 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) or Akyatan (Turkey). 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 PVA, 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. 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.

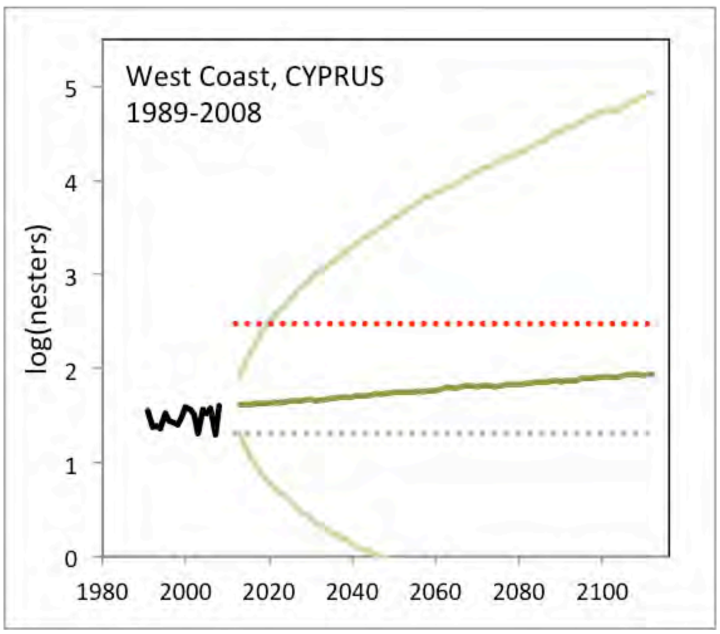

Figure 6.3. Stochastic Exponential Growth (SEG) Model Output for West Coast, Cyprus (Region B only; Demetropoulos and Hadjichristophorou, 2010). Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, gray-green dotted line 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).

For this population, the outputs of the PVA model based on 20 years (1989-2008; Demetropoulos and Hadjichristophorou, 2010) 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 will fall 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 relatively low population sizes of the nesting sites. Within the Mediterranean, rookeries are characterized by one dominant haplotype CM-A13 and a recent study showed no population substructuring between several rookeries in Cyprus and Turkey (Bagda *et al.*, 2012). However, analysis using unpublished data from additional rookery samples in Cyprus shows evidence for two stocks: (1) Cyprus (Karpaz, North Cyprus and Lara Bay; Bagda *et al.*, 2012, Dutton unpublished) and (2) Turkey (Akayatan, Alata, Kazanli, Samandag and Yumurtalık; 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; Broderick *et al.*, 2007). Post-nesting females migrate primarily along the coast from their nesting beach to foraging ground, increasing likelihood of interacting with fisheries (Broderick et. al., 2002a).

The demography of green turtles in the Mediterranean appears to be consistent among the various nesting assemblages (Broderick and Godley, 1996; Broderick *et al.,* 2002a). 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 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 is 12.5 days at Alagadi, Northern Cyprus (Broderick *et al*., 2002a).

6.2.4. 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.

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: Kasparek *et al*., 2001). The nesting season is consistent throughout this DPS (June to August; Broderick *et al.*, 2002a), thus limiting the temporal buffering against climate change in terms of impacts due 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.

All of these factors were considered given current management regimes. 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, Kazanli, 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 Kazanli beach, nesting habitat has been degraded by the construction of greenhouses in the dunes (Kasparek *et al.*, 2001; Türkozan and Kaska, 2010). Coastal construction on Samandağ and Kazanli 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 Kazanli beach in Turkey, light pollution from a soda-chrome factory, the town, and tourist facilities affects the quality of the nesting habitat, as well as threatens nesting females and hatchlings (Kasparek *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 Kazanli 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 Türkozan 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). Although Akyatan beach in Turkey is located in a relatively remote location, there is concern that a substantial amount of tourists, visiting beaches during the summer months may enter this protected beach from around the Tuzla area (Kasparek *et al.*, 2001; Turkozan and Kaska, 2010). Kazanli beach 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).

Neritic/Oceanic Zones

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 affecting 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 affect 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 (Camiñas, 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 Kazanli 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 fisherman in western Cyprus annually. Similar taking of green turtles by fishermen in Greece has also been reported. Some turtles, mostly loggerheads, exhibit head traumas attributed to intentional hits after incidental capture in fishing gear (Panagopoulos *et al.*, 2003; 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 affect 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 5 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).

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. The analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels. 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, 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 Tyrrhenian 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 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. 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 (Camiñas, 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 affect 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 (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 (Camiñas, 2004). Males may also be affected 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 Keratsiili, 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 (Camiñas, 2004; Margaritoulis, 2007). Monofilament netting appears to be the most dangerous waste produced by the fishing industry (Camiñas, 2004).

Contaminants in the marine environment may affect green turtles, although not to the extent they are likely to affect 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 Kazanli nesting beach in Turkey is cause for concern (Kasparek *et al.*, 2001; Venizelos and Kasparek, 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 (Kasparek *et al.*, 2001; Camiñas, 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 hatchling 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.

6.2.5.5.6. 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 hatchling 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. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

In Turkey, three important green turtle nesting beaches (Alata, Kazanli, 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 (Kasparek *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 (Kasparek *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

Patara) used for nesting by green turtles were designated Specially Protected Area status and one beach (Akyata) was designated a Wildlife Reserve (Margaritoulis, 2007).

6.2.6.2. International Instruments

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 Convention on the Law of the Sea
- 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.

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

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 quasi-extinction under current management regimes 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.

	Probability of Reaching Quasi-Extinction					
	<1%	**1–5%**	**6–10%**	**11–20%**	**21–50%**	**>50%**
MEAN ASSIGNED POINTS	10.1	11.7	17.6	27.9	23.9	8.7
SEM	4.9	3.8	4.2	7.8	6.4	4.2
Min	0	0	0	0	0	0
Max	50	30	50	80	80	50

Of the six categories describing the probability that the Mediterranean DPS will reach quasi-extinction 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 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 affect this DPS. The SRT considered these threats heavily in the overall extinction risk analysis.

The SRT determined the likelihood of reaching quasi-extinction 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 at the border of Panama and Colombia at 77° W, 7.5° N, heads due north to77° W, 10.5° N, then northeast to 63.5° W, 19° N, and along 19° N latitude to Mauritania in Africa, to include the U.S. Virgin Islands in the Caribbean. It extends along the coast of Africa to South Africa, with the southern border being the 40° S latitude. Green turtle nesting occurs on beaches along eastern South America from Brazil to the Caribbean portion of the South Atlantic including Caribbean South America, along the western coast of Africa from mid-Mauritania to South Africa, and in the middle of the South Atlantic on Ascension Island (Figure 7.1).

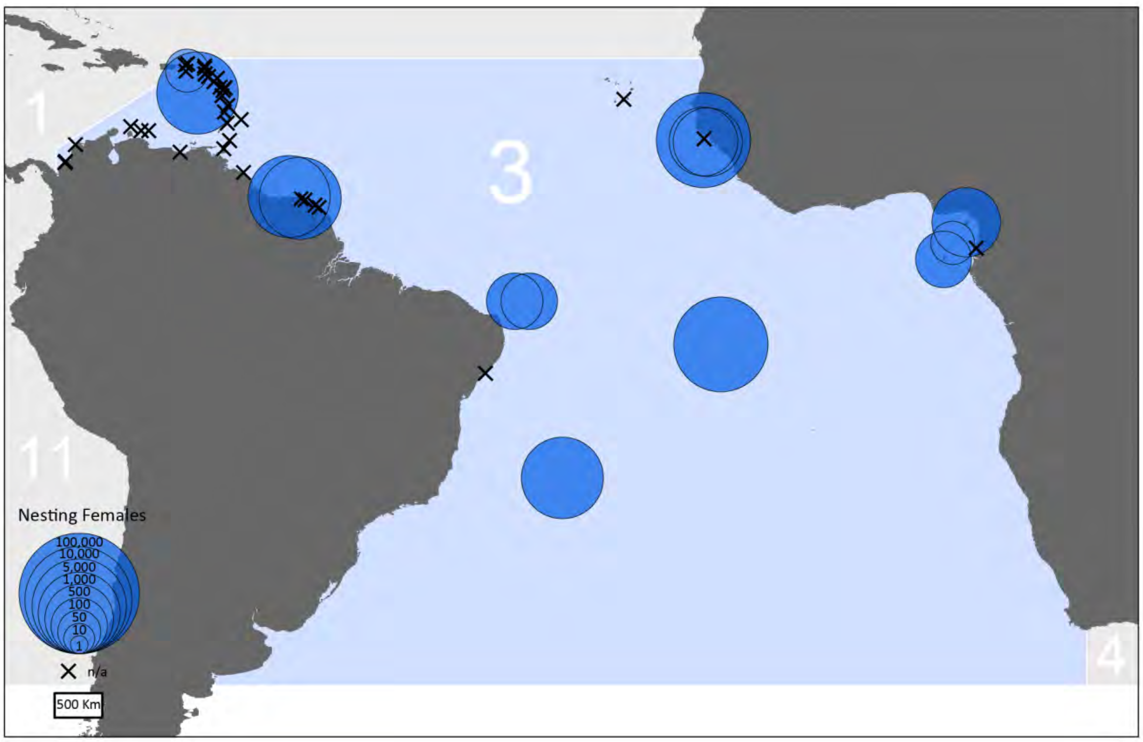

Figure 7.1. Nesting distribution of green turtles in the South Atlantic (blue-shaded area). Size of circles indicates nesting abundance category. Locations marked with 'x' indicate nesting sites lacking abundance information.

The South Atlantic DPS nesting sites can be roughly divided into four regions: western Africa, Ascension Island, Brazil, and the South Atlantic Caribbean (including Colombia, the Guianas, and Aves Island in addition to the numerous small, insular nesting sites). In the eastern South Atlantic, primary green turtle nesting beaches are found along the west coast of the African continent including Bioko Island, Equatorial Guinea; the Bijagos Archipelago, Guinea Bissau (including the largest nesting site in the DPS at Poilão); and São Tome and Principe, with scattered, limited nesting on other insular and mainland beaches. Ascension Island, UK is the only green turtle nesting site in the central South Atlantic. In the western South Atlantic there are important rookeries off Brazil, on Trindade Island, Atol das Rocas, and Fernando de Noronha, with smaller rookeries also occurring on the Brazilian mainland coast. 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.

The in-water range of the South Atlantic DPS is similarly widespread. In the eastern South Atlantic, significant sea turtle habitats have been identified, including green turtle feeding grounds in Corisco Bay, Equatorial Guinea/Gabon (Formia, 1999); Congo; Mussulo Bay, Angola (Carr and Carr, 1991); as well as Principe Island. 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, 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 (Lopez-Mendilaharsu *et al.,* 2006, Lezama, 2009; González Carman *et al.*, 2011; Prosdocimi *et al.*, 2012; Rivas-Zinno, 2012).

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.

7.1.1. Nesting Abundance

For the South Atlantic DPS, we identified 51 total nesting sites. Of those sites, some are individual beaches while others epresent multiple nesting beaches, 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; Antiqua 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, Farez, Irakumpapi, Organabo (French Guiana); Kourou and Karouaba beaches (French Guiana); Cayenne-Montjoly (French Guiana); Guadaloupe; Petite Terre-Terre de Bas (Guadaloupe); Petite Terre-Terre de

Haut (Guadaloupe); Les Galets de Marie-Galante (Guadaloupe); 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 Sauipe, 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 Poilao; 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.

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
United Kingdom	Ascension Island	2010–2012	13,417
Brazil	Atol das Rocas	2005–2008	275
Brazil	Fernando de Noronha	2008–2012	70
Brazil	Trindade Island	2008–2010	2,016
Venezuela	Aves Island	2010	2,833
Suriname	Matapica Reserve	2008–2010	3,661
Suriname	Galibi Reserve	2008–2010	9,406
United States (USVI)	Buck Island	2006–2007	63
Equatorial Guinea	Bioko	2002/2003 and 2004/2005	850
São Tomé and Principe	Praia Grande	2007/2008 and 2009/2010	300
São Tomé and Principe	Principe	2009	76
Guinea-Bissau	João Vieira	2011	596
Guinea-Bissau	Orango National Park	1992–1993	513
Guinea-Bissau	Poilão (Bijagos Archipelago)	2007	29,016*

* 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.

NESTER ABUNDANCE	# NESTING SITES DPS 3
unquantified	37*
1–10	0
11–50	0
51–100	2
101–500	3
501–1000	3
1001–5000	3
5001–10000	1
10001–100000	2**
>100,000	0
TOTAL SITES	**51**
TOTAL ABUNDANCE	**63,332**
PERCENTAGE at LARGEST NESTING SITE	**46 % (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.

7.2.2. 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.

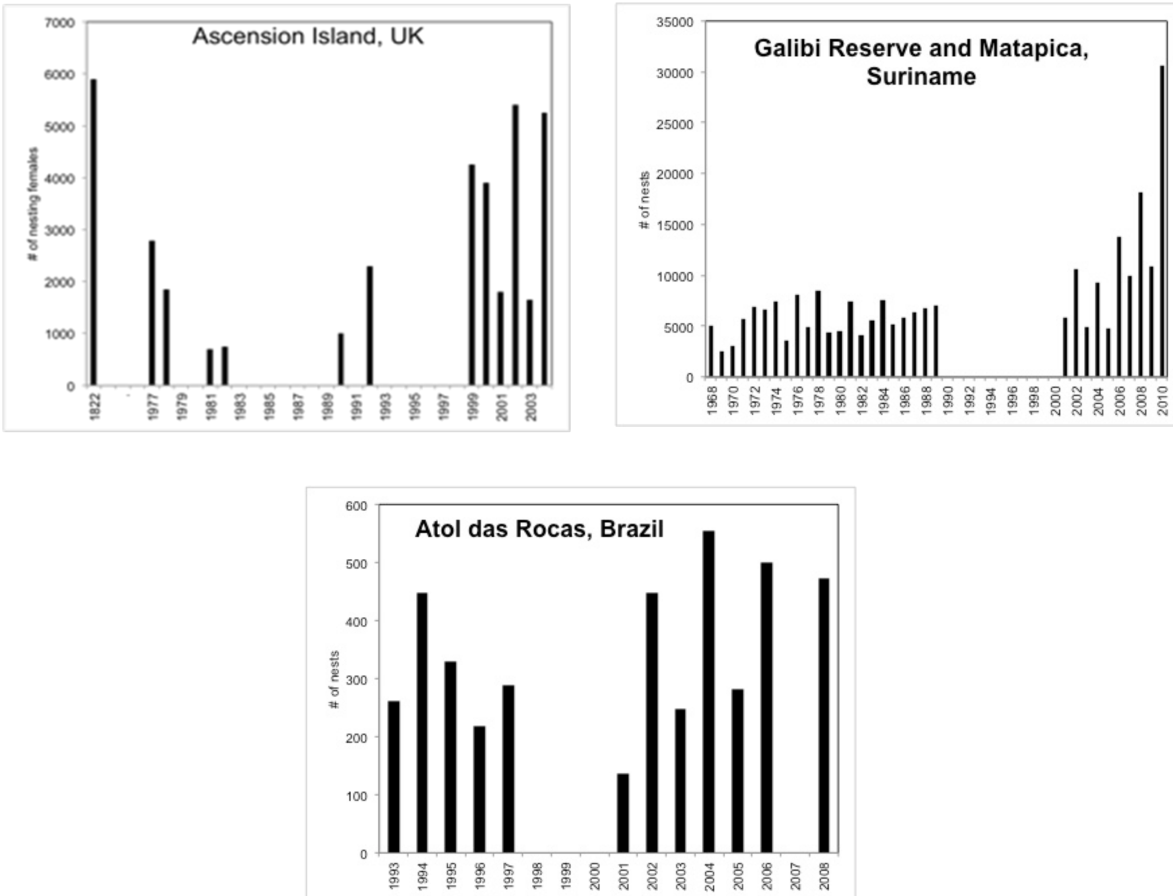

Figure 7.2 Abundance data for green turtle nesting in the South Atlantic DPS with greater than 10 years of recent monitoring data, although with some missed years. These sites are Ascension Island (12 yrs, not counting the 1822 monitoring), Atol das Rocas, Brazil (17 yrs), and Galibi Reserve and Matapica (two 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.

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 (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.

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 Trindade Island, Brazil. This nesting population has been stable with a mean of approximately 1,500–2,000 females nesting per year since the early 1980s (Moreira *et al.*, 1995; Moreira and Bjorndal, 2006; Almeida *et al.*, 2011). In Fernando de Noronha, despite no data having been published yet, nesting numbers are increasing; the average in 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 (A. J. B. Santos, TAMAR, pers. comm., 2014).

At Aves Island, Venezuela, the population has increased steadily. 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 nests per 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 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 per searson were deposited (approximately 500 females per season; 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).

7.2.3 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 phylogenic relationship of the eastern Caribbean rookeries indicates 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. 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; Shamblin *et al.*, 2012a).

In the Southwest Atlantic, foraging areas in Brazil are mainly made up of turtles from Ascension Island, Trinidad and, to some degree, Suriname (Naro-Maciel *et al.*, 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 Guinea Bissau (19 percent) suggesting that, like the 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). 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. 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 portion of the North Atlantic DPS, e.g. Suriname, Costa Rica and Puerto Rico, and Brazil (Ceará) in the South Atlantic DPS has been established by flipper tag recoveries (Godley *et al.*, 2003b; 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 (1997) 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).

7.1.4. 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.

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 South Atlantic 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 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 Island are also relatively common in African foraging grounds.

7.2.5. 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.

All of these factors were considered given current management regimes. 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.

7.2.5.1 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 on oceanic islands which are marine protected areas, females, eggs and hatchlings are fully protected (Almeida *et al.*, 2011a; Bellini *et al.*, 2013). At continental sites destruction and modification of sea turtle nesting habitat (for green turtles and other species) 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 Marczwski, 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 the island (Bellini *et al.*, 2013). In Fernando de Noronha Archipelago, despite the small number of nests (from 55 to 134 for the last five years; N. Marcovaldi, Projeto TAMAR, unpublished data, 2014) the nesting area is fully protected by two federal marine protected areas, the Environmental Protection Area since 1986 and also the Marine National Park, since 1988.

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 (Fretey, 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)). 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 (Valdéz-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.

7.2.5.2. 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 dei Marcovaldi, 1999; Marcovaldi *et al.*, 2005); Almeida and Mendes, 2007). As an exception, in Ceará there are records of illegal commerce and consumption of green turtle meat (E. Lima, Projeto TAMAR,pers. comm., 2014). Use of sea turtles, including green turtles, for medicinal purposes occasionally occurs in northeastern Brazil (Alvez and Rosa, 2006; Braga-Filho and Schiavetti 2013). 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 controlled harvest was allowed until the early 2000s via permit. 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; Humber *et al.*, 2014).

7.2.5.3. 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 0 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 Principe 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 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 Poilão, 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 Poilão 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 (*Dusycion vetulus*; 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; *Canus* 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 (Bornatowski *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 (*Felus* sp.) and dogs (*Canus* 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.

7.2.5.4. 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 South Atlantic green turtles and impacts to their habitats, these regulatory mechanisms are insufficient or are not being implemented effectively to protect green turtles. The analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels. 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.

7.2.5.5. 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 the juvenile stage (Barata *et al.*, 2011).

Similarly, artisanal gill net fisheries in the coastal waters of the Rio 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 affecting 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. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

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 al.*, 2007) 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. 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 such 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, 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 promotes 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.prictma.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).

7.2.6.1. 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 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

British Virgin Islands

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 prohibits the capture of sea turtles from the 1st 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.

Mauritania

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

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 percent 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 (Fretey, 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 alllows 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.

7.2.6.2. 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 Nations Convention on the Law of the Sea
- United States Magnuson-Stevens Fishery Conservation and Management Act
- United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing
- 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

- Council Regulation (EC) No. 1239/98 of 8 June 1998 Amending Regulation (EC) No. 894/97 Laying Down Certain Technical Measures for the Conservation of Fishery Measures (Council of the European Union)

7.3 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 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.

7.4. 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.

	Critical Assessment Elements					
	Element 1	Element 2	Element 3	Element 4	Element 5	Element 6
	Abundance (1 to 5)	Trends / Productivity (1 to 5)	Spatial Structure (1 to 5)	Diversity / Resilience (1 to 5)	Five-Factor Analyses (0 to –2)	Conservation Efforts (0 to 2)
MEAN RANK	1.58	1.92	1.33	1.67	-0.83	0.75
SEM	0.19	0.15	0.14	0.14	0.11	0.22
RANGE	1–3	1–3	1–2	1–2	(–1)–0	0–2

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 did 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 quasi-extinction under current management regimes 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.

	Probability of Reaching Quasi-Extinction					
	<1%	**1-5%**	**6-10%**	**11-20%**	**21-50%**	**>50%**
MEAN ASSIGNED POINTS	69.00	16.50	9.92	4.17	0.42	0.00
SEM	9.05	3.43	4.18	2.29	0.42	0.00
Min	10	2	0	0	0	0
Max	98	40	45	20	5	0

Of the categories describing the probability that the South Atlantic DPS will reach quasi-extinction 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.

7.5. 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 quasi-extinction 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 other ongoing threats known to affect the DPS. Further, 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 in the U.S. Virgin Islands 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.

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, 2012; Mortimer 1984; Table 8.1).

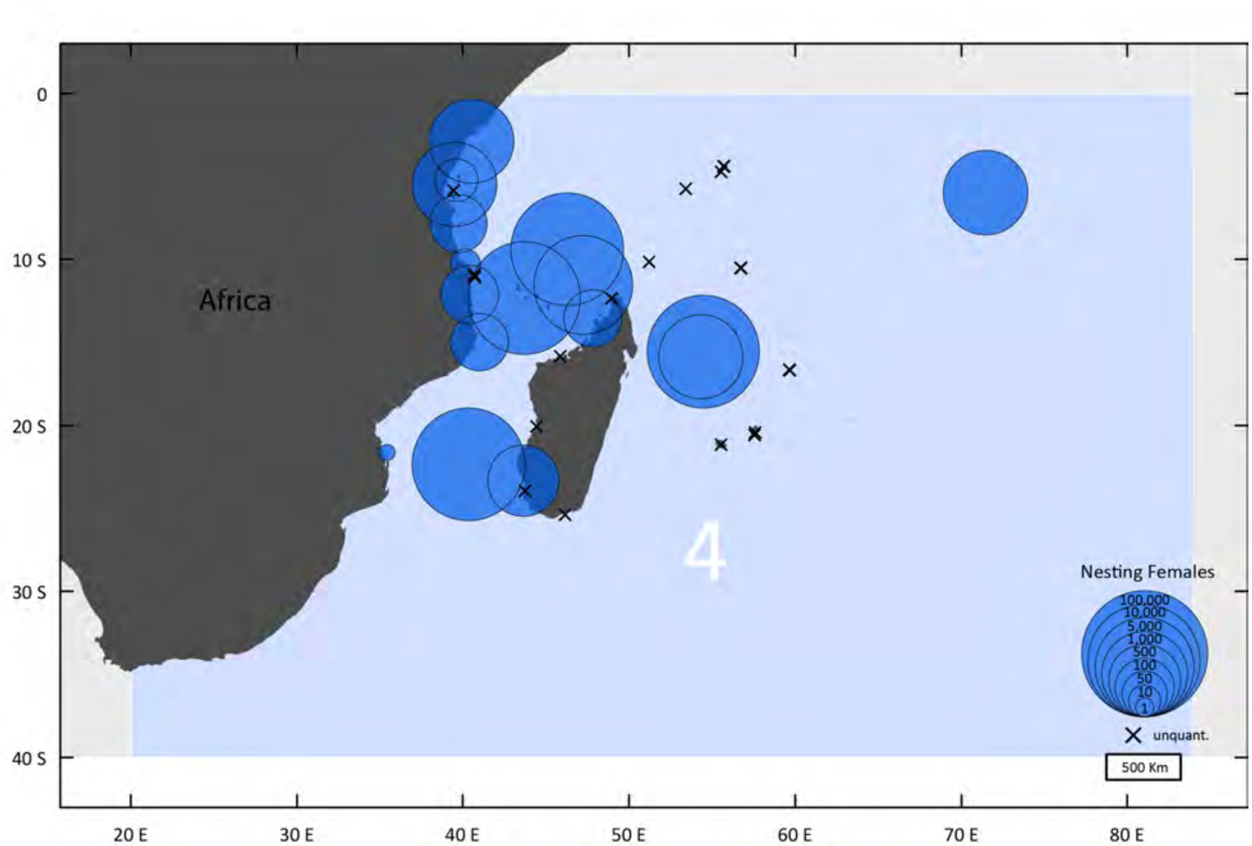

Figure 8.1. Nesting distribution of green turtles in the Southwest Indian DPS. Size of circles indicates estimated nester abundance (see Section 8.2.1). Locations marked with '×' indicate nesting sites lacking abundance information

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 in Mozambique 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 20m 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, four of which have more than 10,000 nesters (~30 percent of the total adult females; Table 8.2).

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.

For the Comoros Islands (Grande Comore, Mohéli, and Anjouan), monitoring of green turtle nesting is conducted at Mohéli Island. 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 a French Island 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 2,239 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 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, 2012). 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 [(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, where given, 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.

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Republic of Seychelles	Aldabra Atoll	2004-2008	16,000
Republic of Seychelles	Assumption, Cosmoledo, Astove, Farquhar	Years not provided	~2,000 females nesting annually
Republic of Seychelles	Amirantes Group	Years not provided	~300 females nesting annually
Republic of Seychelles	Inner Islands	Years not provided	~50 females nesting annually
Islamic Republic of Comoros	Mohéli	2000–2007	15,000
France (Indian Ocean)	Mayotte	1998–2006	12,000
France (Indian Ocean)	Tromelin	1987–2006	4,500
France (Indian Ocean)	Europa	1984-2006	25,500
France (Indian Ocean)	Glorieuses	1987–2006	6,000
Madagascar	Nosy Iranja Kely	2003	153
Kenya	Entire Coastline	2000	1,500
Mozambique	Coastline and Islands	2004–2012	150
Tanzania (including Zanzibar)	Zanzibar: Pemba, Unguja, Mnemba, Misali Islands	Sporadic since early 1990s	1,500
British Indian Ocean Territory (Mauritius)	Chagos Archipelago	1996, 1999, 2006	1,800
France (Indian Ocean)	La Reunion Island	2004-2005	6

Table 8.2. Green turtle nester abundance distribution among nesting sites in the Southwest Indian DPS.

NESTER ABUNDANCE	# NESTING SITES DPS 4
unquantified	23*
1–10	1
11–50	0
51–100	0
101–500	3
501–1000	1
1001–5000	4
5001–10000	2
>10,000	4
TOTAL NESTING SITES	37
TOTAL ABUNDANCE	91,059
PERCENTAGE AT LARGEST NESTING SITE	**30% (Europa, Eparses Islands)**

* Not included in Table 5.1

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.

The green turtle shows overall large, stable or increasing nesting populations in the French Eparses Islands and Mayotte. Protected nesting sites with long-term monitoring indicate that five out of six sites have shown stable or increasing abundance (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, Europa and Tromelin, Eparses Islands with 19, 20, and 20 yrs, respectively; Figure 8.2). 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).

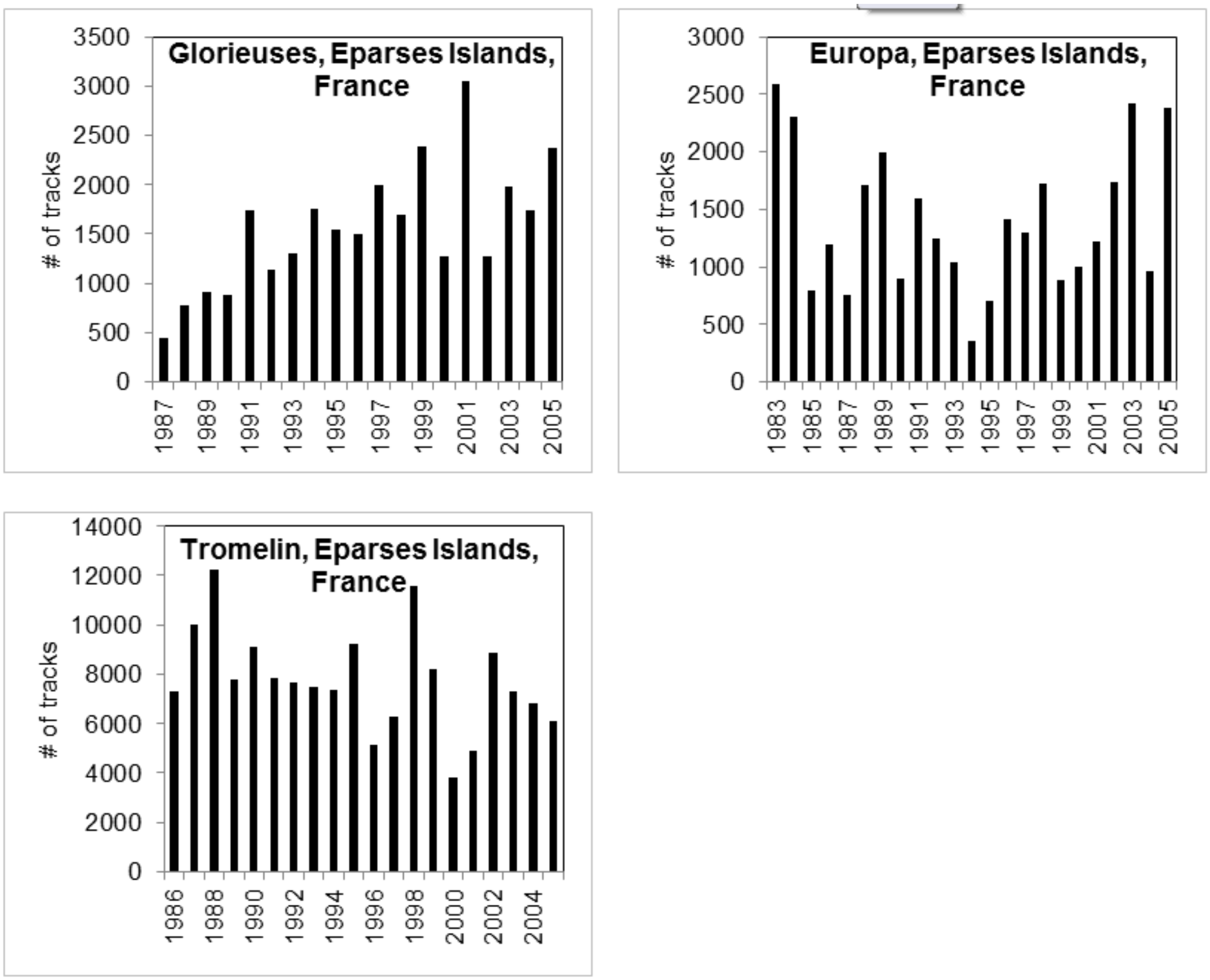

Figure 8.2. Trend data for green turtle crawls in the Southwest Indian DPS for sites with greater than 10 years of recent monitoring data. These are Glorieuses, Eparses Islands (19 yrs); Europa, Eparses Islands (23 yrs); and Tromelin, Eparses Islands in France (20 yrs). Data represents tracks collected on 16 percent of the total beach on Grande Glorieuse, 26 percent on Europa, and 100 percent on Tromelin.

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 Lebeau *et al.* (1983). 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, as shown in Figure 8.2 (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.

At Glorieuses, French Eparses Islands, annual growth rate was 3.5 percent from 1987 to 2008 (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 Tromelin, Eparses Islands, France, annual growth rate was -1.7 percent from 1986 to 2008 (Le Gall *et al.*, 1986; Lauret-Stepler *et al.*, 2007; Bourjea, 2012). At Mayotte, the annual growth rate was 0.9 percent from 1998 to 2006 (Bourjea *et al.*, 2007b; Bourjea, 2012).

While true trends cannot be ascertained in many cases for other sites due to the lack of data, we discuss the indications of possible trends at some of the other primary nesting sites.

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). Since 2000, there have been 4,000-6,000 nesters per year (Bourjea, 2012).

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 fewer 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 6,038-8,734 nests/yr during 1981 to 1984 to about 15,670 nests/yr from 2004 to 2008 (Mortimer *et al.*, 2011). 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 Island of the Comoros Islands, the annual population growth rate was 20 percent from 2000 to 2007 (Bourjea, 2012).

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 Reunion Island, 11 green turtle nests, likely representing 4 turtles, were recorded in a recent 2-year period (Ciccione and Bourjea, 2006).

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* *al.*, 2006). The authors suggested that the SMC stock 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 ($F_{ST} = 0.3030$). 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 along mid-oceanic sea grass beds. This behavior is believed to be mainly attributable to the fact that 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. The median CCL of nesters at Mayotte Island from 1998 to 2005 was 108 cm (Bourjea *et al.*, 2007a). The internesting 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 Mohéli, 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 percent +/- 33.37 percent (Innocenzi *et al.*, 2010).

8.2.4. Diversity / Resilience

The components considered under this critical assessment 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 are made of coralline sand. Nesting occurs throughout the year with peaks that vary among 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.

All of these factors were considered given current management regimes. 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 has been reported for Comoros Islands (Ahamada, 2008; Bourjea, 2012); Mozambique (Costa *et al.*, 2007; Videira *et al.*, 2008); Tanzania (Bourjea, 2012); Madagascar (Ciccione *et al.,* as cited in Bourjea, 2012; Lilette, 2006 as cited in Bourjea, 2012; Rakotonirina and Cooke, 1994); and Kenya (Bourjea, 2012). Egg exploitation 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 over on the nesting beach. A similar decrease has been reported for Aldabra, where green turtles were heavily exploited until 1968, which is located just 20 miles from Assumption Island (Mortimer, 1984). After 1968; however, green turtles have been protected at Aldabra (J. Mortimer, Seychelles Dept. of Evironment, 2014).

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; Lilette, 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.

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.

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.

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). The analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels. 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 (Peterson, 2005; Louro *et al.*, 2006; Costa *et al.*, 2007; Fennessy and Isaksen, 2007; 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. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

The international regulatory mechanisms described in Section 8.2.6.2 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.

Nine countries of the southwest Indian Ocean conceived and signed the IOSEA (www.ioseaturtles.org): Comoros in June 2001, United Republic of Tanzania in June 2001, Kenya in May 2002, Mauritius in July 2002, Madagascar in January 2003, Seychelles in January 2003, South Africa in February 2005, Mozambique and France (Indian Ocean) in December 2008. IOSEA aims to develop and assist countries of the region in the implementation of the IOSEA regional strategy for management and conservation of sea turtles and theirs habitats. Accordingly, IOSEA has been successfully coordinating and closely monitoring region-wide conservation efforts in the Indian Ocean for years. This has included the development of a state-of-the-art online reporting facility, satellite tracking, genetic regional database, flipper tag inventory and a global bibliographic resource.

Also within the Southwest 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 Tanzania, as well as representatives from intergovernmental organizations, academic, and non-governmental organizations within the region.

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 neasting 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-Econimique d'Itsamia has been preserving the importance of the beaches here for green turtles through protective actions since 1994 (Innocenzi *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
- Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region
- Ramsar Convention on Wetlands
- United Nations Convention on the Law of the Sea
- 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.

	Critical Assessment Elements					
	Element 1	Element 2	Element 3	Element 4	Element 5	Element 6
	Abundance (1 to 5)	Trends / Productivity (1 to 5)	Spatial Structure (1 to 5)	Diversity / Resilience (1 to 5)	Five-Factor Analyses (–2 to 0)	Conservation Efforts (0 to 2)
MEAN RANK	1.25	1.75	1.42	1.58	–0.75	0.75
SEM	0.13	0.13	0.15	0.15	0.22	0.18
RANGE	1-2	1-2	1-2	1-2	(–2)-0	0-2

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 quasi-extinction under current management regimes 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.

	Probability of Reaching Quasi-Extinction					
	<1%	**1–5%**	**6–10%**	**11–20%**	**21–50%**	**>50%**
MEAN ASSIGNED POINTS	71.33	13.58	7.58	5.50	2.00	0.00
SEM	9.20	3.76	2.84	4.17	1.66	0.00
Min	0	1	0	0	0	0
Max	99	50	30	50	20	0

Of the six categories describing the probability that the Southwest Indian DPS will reach quasi-extinction 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). However, our analysis did not consider the scenario in which current laws or regulatory mechanisms were not continued. Given the conservation dependence of the species, without mechanisms in place to continue conservation efforts in this DPS, some threats could increase and population trends could be affected.

9. NORTH INDIAN DPS (DPS #5)

9.1. DPS Range and Nesting Distribution

The North Indian DPS begins at the border of Somalia and Kenya north into the Gulf of Aden and the Red Sea tothe Persian Gulf and east to the Gulf of Mannar off the southern tip of India and including a major portion of India's southeastern coast up to Andra Pradesh. The southern and eastern boundaries are the equator (0°) and 84°E, respectively, which intersect at the equator and 84°E in the southeast corner of the DPS. It is bordered by the following countries (following the water bodies from west to east): Somalia, Djibouti, Eritrea, Sudan, Egypt, Israel, Jordan, Saudi Arabia, Yemen, Oman, United Arab Emirates, Qatar, Bahrain, Kuwait, Iraq, Iran, Pakistan, India, and Sri Lanka.

Nesting is concentrated primarily in the northern and western region of the North Indian DPS from the Arabian Peninsula to the Pakistani-Indian border, with smaller but significant nesting colonies occurring in Sri Lanka, India's Lakshadweep Island group, and the Red Sea (Figure 9.1). Nesting in the Arabian Gulf occurs in low numbers.

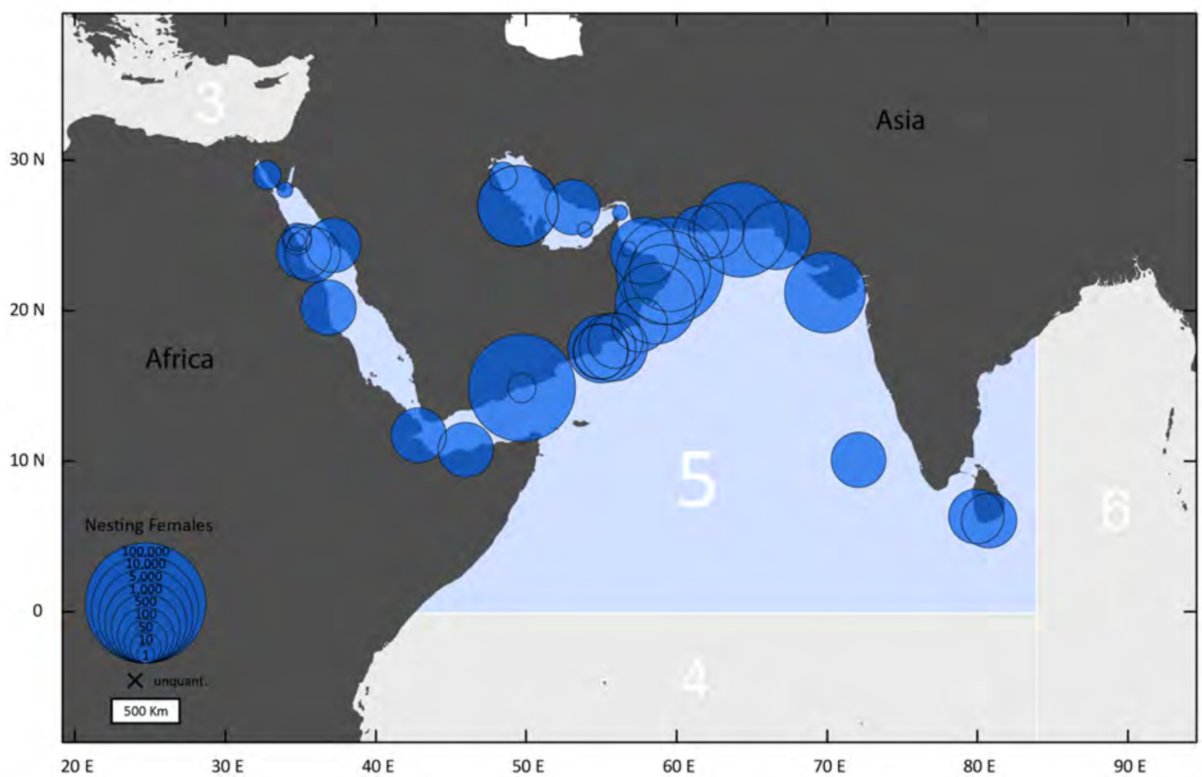

Figure 9.1. Nesting distribution of green turtles in the North Indian DPS (blue-shaded area labeled '5'). Size of circles indicates estimated nester abundance (see Section 9.2.1).

Seagrass beds are extensive within the DPS, although a comprehensive understanding of juvenile and adult foraging areas is lacking. There are extensive foraging areas in the Arabian Gulf, on the coasts of Oman and Yemen, Gulf of Aden, and in the Red Sea (Ross and Barwani, 1982; Salm, 1991; Salm and Salm, 2001). Barr al Hickman, along the Sahil al Jazit coastline in Oman, represents one of the most important known foraging grounds for green turtles. Foraging areas on the coasts of the Arabian Sea are extensive (Jupp *et al.*, 1996 as cited in Ferreira *et al.*, 2006). Juvenile green turtles have been sighted and captured year-round in the lagoons in Agatti and Kavaratti. These Lakshadweep lagoons are known to be important developmental habitat for green turtles in this DPS (Tripathy *et al.*, 2002, 2006).

9.2. Critical Assessment Elements

In the evaluation of extinction risk for green turtles in the North 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. See Section 3.3 for additional information on the selection of these six critical assessment elements.

9.2.1. Nesting Abundance

For the North Indian DPS, we identified 38 total nesting sites. Of those sites, some are individual beaches while others may be multiple nesting beaches lumped together. Nesting abundance data for this DPS are available at nesting areas in Sri Lanka, India, Pakistan, Iran, Kuwait, Oman, Yemen, Saudi Arabia, Egypt, Djibouti, and Somalia (Table 9.1). Among the nesting sites with adult female estimates, the largest known nesting site, Sharma, Yemen accounts for almost 31.6 percent of the total females (Table 9.2).

Table 9.1. Summary of green turtle nesting sites in the North Indian 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.

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Djibouti	Djibouti	2004	300
Egypt	Egypt (Wadi Al-Gimal, Ras Banas, Sarenka, Siyal, and Rowabill Island)	2012	156
Egypt	Ras Bagdadi	2001-2006	15
Egypt	Ras Honkorab	2011	4
Egypt	Ras Shartib	1967	40
Egypt	Sharm El-Sheikh	2011	1

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Egypt	Umm Al-abass	2011	13
Egypt	Zabargard Island	2012	444
Eritrea		*	n/a
India	Gujarat	1999 (low), 2004 (high)	1,125
India	Suheli Island, Lakshadweep Islands Group	2012	225
Iran	Iran	2003	30[1]
Kuwait	Qaru	2008 - 2011	16
Oman	Al Halaniyat Islands	**	750
Oman	Batinah	1990	9
Oman	Daymaniyat Islands	**	900
Oman	Hasik to Ra's Hasik	**	619
Oman	Musandam Island	1990	9
Oman	Masirah Island	1977 (low) 1986 (high)	1,125
Oman	North coast of Ras Al-Hadd	**	1,875
Oman	Ras Al-Hadd	1977-1986; 2007	16,184
Oman	Ra's Jifan to Ra's Jibsh	2000	1,500
Oman	Ra's Madrakah area	**	375
Oman	Ra's Nuss	**	188
Oman	South of Hadbin	**	127
Oman	Sharbithat area	**	210
Yemen	Ras Sharma	***	18,000
Pakistan	Gwadar and Pasni	1997 (less than a week survey window)	300
Pakistan	Hawkes Bay and Sandpit	1997	720
Pakistan	Daran Beach, Jiwani	1999-2008	371

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Pakistan	Kamgar Beach at Ormara	1986-1987	6,000
Somalia		***	150
Sudan		***	150
Saudi Arabia	Saudi Arabian Gulf Islands (Jana, Karan, Juraid and Kurayn)	1986-1997	2,410[2]
Saudi Arabia	Ras Baridi	2003	165
Sri Lanka	Rewaka Beach	2006-2011	455
Sri Lanka	Kosgoda	2003 (low) and 2006 (high)	281
Sir Bu Nair Island	United Arab Emirates	2012	1

[1] This number was updated based on external review subsequent to SRT voting. While an appreciable change for this particular site (from 1,500 to 30), we don't believe the revision would appreciably change how the SRT evaluated risk for the DPS.
[2] This number was changed based on external review subsequent to SRT voting. Although the number changed from 1,590 to 2,410, we don't believe the revision would appreciably change how the SRT evaluated risk for the DPS.
*No year available but reference Goitom *et al.*, 2006
**No year available but reference Salm *et al.*, 1993
***No year available but reference Persga/GEF, 2004

Table 9.2. Green turtle nester abundance distribution among nesting sites in the North Indian DPS.

NESTER ABUNDANCE	# NESTING SITES DPS 5
unquantified	1
1-10	5
11-50	5
51-100	0
101-500	15
501-1000	4
1001-5000	5
5001-10000	1
10001-100000	2
>100,000	0
TOTAL SITES	**38**
TOTAL ABUNDANCE	**55,243**
PERCENTAGE at LARGEST NESTING SITE	**33% (Ras Sharma, Yemen)**

9.2.2. Population Trends

Although there are no sites for which there is long-term published data based on standardized surveys that can be used for a PVA, nine nesting sites are examined in the North Indian DPS using the best available data: Daran Beach, Jiwani (Pakistan), Zabargard Island (Egypt), Ras al Hadd (Oman), Hawkes Bay and Sandspit (Pakistan), Gujarat (India), Ras Sharma (Yemen), Karan and Jana Island (Saudi Arabia), Juraid Islands (Saudi Arabia), and Rewaka Beach (Sri Lanka). Daran Beach, Jiwani (Pakistan) and Zabargard Island (Egypt) are the only 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 9.2). It is difficult to ascertain trends from these data. For a list of references on trend data, see Appendix 3.

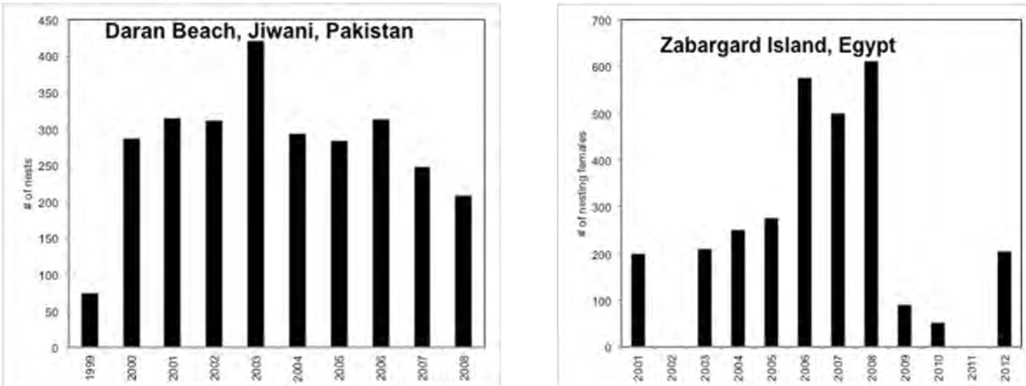

Figure 9.2. Nesting data for green turtle nesting sites in the North Indian DPS with 10 or more years, although with some missed years. These sites are Daran Beach, Pakistan and Zabargard Island, Egypt.

Oman remains one of the most important nesting concentrations of green turtles in the Indian Ocean, if not the entire world. At Ras al Hadd (Oman), Ross and Barwani (1982) reported approximately 6,000 females nesting each year for the period 1977 to 1979, and Groombridge and Luxmoore (1989) described the same number for the late 1980s. Although annual nesting totals have not been published since the 1980s, monitoring in the mid-to-late 1990s and early 2000s by park rangers indicate that nesting during peak periods ranges from 200-400 females per night (AlKindi *et al.*, 2003), and approximately 44,000 nests were recorded in 2005 for Ras al Hadd and Ras al Jinz nesting beaches (S. Al-Saady, Ministry of Regional Municipalities, Environment and Water Resources, pers. comm., 2007).

At the other six sites, the level of year to year standardization is unknown and therefore the usefulness of these data for understanding trends is questionable at best. With this caveat, however, declines are evident at Hawkes Bay and Sandspit (Pakistan), where a mean of approximately 1,300 nests were deposited annually from 1981 to 1985 (Groombridge and Luxmoore, 1989) and a mean of approximately 600 nests were laid from 1994 to 1997 (Asrar, 1999). At Gujarat (India), 866 nests were deposited in 1981 (Bhaskar, 1984) and 461 nests in 2000 (Sunderraj *et al.*, 2006a); however, because there are only two data points, it is not possible to determine a trend. At Ras Sharma, counts of nightly nesters during peak nesting season in 1966 and 1972 (30-40 females; Hirth, 1968; Hirth and Hollingsworth, 1973)) versus the same index during the peak of the 1999 nesting season (15 females; Saad, 1999) are suggestive of a decline. Again the lack of multiple-year data sets for both Gujarat and Ras Sharma preclude trend assessment. This is particularly true since Saad (1999) only worked at one beach predominantly, while estimates from Hirth (1968) and Hirth and Hollingsworth (1973) represented a greater area (N. Pilcher, Marine Research Foundation, pers. comm., 2007).

In Saudi Arabia, data are available for eight seasons from Karan and Jana Islands (1986 and 1997) and only a single season for Juraid Islands (1991), indicating that approximately 600 nests are deposited each year between these sites (Pilcher, 2000). At Karan Island alone (Saudi Arabia), 500-1,000 females nested annually during the 1970s (Basson *et al.*, 1977), and during the 1991 and 1992 seasons, 559 and 408 females nested, respectively (Pilcher, 2000). However, the fact that so few years of nesting data are available suggests that this figure should be used cautiously if attempting to derive an annual mean.

At Rewaka Beach (Sri Lanka), a mean of 184 females nested each year from 1996-2000 (Kapurusinghe, 2006), but as with other short-term data sets, no trend can be established.

No sites in the North Indian DPS met our standards for conducting a PVA (for more on data quantity and quality standards used, see Section 3.2).

9.2.3. Spatial Structure

When examining spatial structure for the North Indian DPS, the SRT examined three lines of evidence, including genetic data, flipper and satellite tagging, and demographic data. The extent of genetic sampling in the North Indian DPS has been limited. Only 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 based on mtDNA analysis. There are no studies of foraging grounds in the North Indian DPS to provide information on the distribution or the mixing of turtles outside of this DPS.

With respect to flipper tagging, a few tag recoveries have been reported with no reported recoveries outside of the North Indian DPS. Satellite tracking was considered. Adult females from Egypt, Sri Lanka, and Oman were tracked during post-nesting migrations. All remained within the North Indian DPS. The satellite telemetry data for nesting females in Sri Lanka provided some information on possible foraging locations which were within the inshore waters of southern Sri Lanka and the Gulf of Mannar Biosphere Reserve, although sample size was limited (Richardson *et al.*, 2013). The satellite telemetry for nesting females in Kuwait verified nesting in Qaru Island. These turtles migrated to the shallow seas in Saudi Arabia (Rees *et al.*, 2013). Satellite telemetry data for two post nesting females in Oman has not been analyzed nor published.

The demography of green turtles in the North Indian DPS appears to vary among the various nesting assemblages. Hatching success varies widely from 39-91 percent for areas with available information (Hirth, 1997; Al-Merghani *et al.*, 2000; Hanafy, 2012). Clutch size varies widely from 41.4 to 122 eggs per nest, and clutch frequency ranges from 1.9 to 5 times per season (Hirth, 1997; Pilcher and Al-Merghani, 2000; Ekanayake and Ranawana, 2001a; Ekanayake *et al.*, 2002, 2013; Rees *et al.*, 2012). Remigration interval varies from 1.4 to 3.5 years by nesting site (Pilcher and Al-Merghani, 2000; Ekanayake *et al.*, 2004, 2010). The estimated age to maturity is 33.3 years at Ras Al-Hadd (Seminoff *et al.*, 2004). The variation in parameters such as remigration interval, clutch size, hatching success, and clutch frequency suggests complexity in the population structuring in the North Indian DPS.

9.2.4. Diversity / Resilience

The components considered under this critical element include the overall nesting spatial range, nester size, 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.

The population is moderately dispersed within the North Indian DPS, although the greatest nesting is concentrated in the northern and western region of the DPS with about 72 percent of the nesting concentrated in Oman and Yemen (see Section 9.1., Nesting Distribution). Nesting female size ranges from 92.6 cm to 106 cm CCL (AlKindi *et al.*, 2003; Mendonça *et al.*, 2010). The nesting season varies widely within the DPS. The peak nesting season in Ras Sharma, Yemen is July. In Gujarat, India the nesting season is from August to March (Sunderraj *et al.*, 2006b), and in Oman, nesting occurs year-round.

No satellite telemetry, tagging, or genetics data suggests dispersal outside North Indian DPS. Mitochondrial DNA studies have only been completed on one stock (Saudi Arabia), and this was found to be very distinct from other rookeries elsewhere in the Indian Ocean. While some level of population substructuring is likely given the spatial distribution and nesting season variation, it cannot presently be confirmed with the limited genetics studies that have been conducted and published.

9.2.5. 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.

All of these factors were considered given current management regimes. The following information on these factors / threats pertains to green turtles found in the North Indian DPS.

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

Coastal development, beachfront lighting, erosion, sand extraction, vehicle and pedestrian traffic consistently affect hatchlings and nesting turtles throughout this DPS. The extent of seagrass degradation is not known but is negatively affected by dredging, pollution, and trophic changes as a result of climate change and occur 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

In the North Indian DPS, some nesting beaches have become degraded from a variety of activities. Destruction and modification of green turtle nesting habitat in the North Indian DPS result from coastal development and construction, beachfront lighting, sand extraction, beach erosion, vehicular and pedestrian traffic, and beach pollution. These activities may directly affect 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 (defined as greater than 100 nests recorded in any year within the past two decades) within this DPS are located in Egypt, Oman, Yemen, Saudi Arabia, India, Pakistan, and Sri Lanka; therefore, the following threats to the nesting populations and habitat are primarily focused on these areas.

One of the largest green turtle nesting population within this DPS is concentrated on the nesting beaches of Ras Al-Hadd, Oman (Ross, 1979). Ras al Hadd, Ras al Jinz and the numerous smaller nesting beaches south of it, are protected from development as part of the Ras al Hadd Nature Reserve. However, upland light pollution is negatively impacting these otherwise suitable nesting habitats. The beaches adjacent to the Town of Ras al Hadd do not have an adequate buffer or topography to protect the nesting turtles and hatchlings from lights behind the beach (E. Possardt, USFWS, pers. comm., 2013). Nesting females and hatchlings are disoriented and misoriented by street lights and lights on various buildings behind this stretch of beach. A wall about 400 m long and 1 m high has been constructed to deter females from wandering away from the nesting beach toward the lights after nesting and to deter hatchling misorientation and disorientation (R. Baldwin, Five Oceans LLC, pers. comm., 2013). The wall has been successful in deterring nesting female wanderings on this short stretch of beach but is an incomplete solution to the problem. Ongoing development in the area to accommodate greater tourism is expected to exacerbate the light pollution problem.

Beach driving by fisherman who haul and launch boats from Ras al Jinz beach in Oman, as well as storage of boats and nets on the nesting beach, is highly problematic and likely decreases the suitability of nesting habitat by creating impediments to nesting and hatchling turtle movements on the beach. However, no assessment has been conducted to determine impacts on nesting turtles and hatchlings (E. Possardt, USFWS, pers. comm., 2013).

Light pollution is increasing near the Karan Island, Saudi Arabia site from oil rig developments but the impact on hatchlings and nesting females is unknown (J. Miller, Biological Research and Education Consultants, pers. comm., 2013). At Ras Baridi, one of the main nesting beaches in Saudi Arabia, uncontrolled particulate emissions from a large cement factory has coated the beaches at times and poses a threat to hatchlings because they are unable to emerge from the nest due to the hardened sand (Pilcher, 1999; PERSGA/GEF, 2004).

The main green turtle nesting beach in Sri Lanka lies within the Rekawa Protected Area and is not subject to beach development threats. However, coastal development and its associated impacts (e.g., artificial lighting, coastal armoring, sand mining, increased human activity) elsewhere in Sri Lanka and Gujarat, India, has been identified as degrading nesting habitat to such an extent that it is no longer suitable for nesting (Kapurusinghe, 2006).

Similarly, the most valuable nesting sites for green turtles along the Red Sea beaches of Egypt are located within the boundaries of the Red Sea Protected Areas (Hanafy, 2012). However, elsewhere along the Egyptian Red Sea, coastal tourism development, human use of beaches, beach reclamation, and coastal lighting threaten sea turtle nesting habitat (Hanafy, 2012;

Mancini, 2012), but it is not possible to quantify the impact of these activities on turtle nesting due to lack of baseline nesting data (Hanafy, 2012).

The beaches of Iran that border the Gulf of Oman are highly erosive and consistently inundated (Mobaraki, 2004). In addition, the green turtle nesting beach at Umm Al-Maradim was lost due to the construction of a large coast guard station in 2005 (Rees *et al.*, 2013).

The most important green turtle nesting beaches in Yemen fall within the Ras Sharma Protected Area, and this nesting habitat is secure from beach development threats.

Neritic/Oceanic Zones

Threats to habitat in the green turtle neritic and/or oceanic zones of the North Indian DPS include fishing practices, marine pollution, and climate change.

Trawling occurs throughout much of the North Indian DPS. This fishing practice has the potential to destroy bottom habitat in these areas. Fishing methods affect neritic zones not only by impacting bottom habitat, including sea grasses that are present, and incidentally capturing turtles, but also by depleting fish populations and thus altering ecosystem dynamics. Bottom trawling is the fishing practice that likely dramatically affects seagrasses. Boat anchoring also may affect green turtle foraging habitat in the neritic environment. Climate change may result in future trophic changes, including changes in the distribution, amount, and types of sea grasses 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. The most dramatic example of the threats to sea turtles and their habitat from oil pollution in the region is the Gulf War oil spill in the Arabian Gulf in 1991, which is estimated to be the largest oil spill in history at the time of this report (ABC, 2010). Indirect effects can result from both point and non-point source pollution associated with coastal development. The impacts of climate change may also result in trophic level alterations, and therefore may affect forage quantity, quality, and/or distribution.

In the Arabian Gulf, extensive seagrass beds provide important foraging sites for green turtles within waters of Bahrain, United Arab Emirates, Qatar, and Saudi Arabia but are being degraded and lost from the continual threat of dredging, siltation, and land reclamation (Pilcher, 2000, 2006; Al-Muraikhi *et al.*, 2005; Abdulqader, 2008; Al-Abdessalaam *et al.*, 2008).

In the waters surrounding the Lakshadweep islands in India, the high densities of green turtles, without the natural level of control from the top predators such as tiger sharks, can cause an increase grazing pressure and reduce the amount of healthy seagrass beds available (N Kelkar *et al.*, 2013; Nachiket Kelkar *et al.*, 2013). Seagrass pastures that have been modified as a result of over-grazing could impact green turtle movement and human response to green turtles as a result of reduced fishing productivity (Lal *et al.*, 2010)

9.2.5.2. Factor B: Overutilization

The harvest of eggs and turtles was likely a factor that contributed to the historical declines of the population. Current harvest of eggs and turtles continues for human consumption and trade in portions of the DPS.

Directed take of eggs by humans occurs at the primary green turtle nesting beaches in Saudi Arabia (Al-Merghani *et al.*, 1996; Pilcher, 2000);Yemen (K. Nasher, Sana'a University, pers. comm., 2013), India (Sunderraj *et al.*, 2006a), and Sri Lanka (Rajakaruna *et al.*, 2009; Turtle Conservation Project, 2009). Directed take of nesting females is also still common at nesting beaches in Yemen (K. Nasher, Sana'a University, pers. comm., 2013). In spite of wildlife protection laws, green turtles are still killed opportunistically for trade in the Bay of Mannar between India and Sri Lanka (Bhupathy and Saravanan, 2006).

An estimated 1,000 green turtles were harvested annually in Oman during the late 1970s (Ross, 1979) and, although sea turtles are considered protected by Royal Decrees and Ministerial Decisions MD 207/93, green turtles and sea turtle eggs are still frequently being harvested for food in Oman (R. Baldwin, Five Oceans LLC, pers. comm., 2013).

Illegal and legal capture of marine turtles and the collection of turtle eggs is rather widespread in the Djibouti and Somalia region of the Gulf of Aden and the Red Sea, and turtle meat, oil and eggs are an important source of subsidiary food for artisanal fishermen (PERSGA, 2001; van de Elst, 2006; Galair, 2009; van de Giessen, 2011; Witsen, 2012).

Harvesting of sea turtle eggs and meat for consumption by local communities and fishermen occurs at a subsistence level in Eritrea (Howe *et al.*, 2004; Pilcher, 2006; Teclemariam *et al.*, 2009), but the pressure on green turtle populations is reported to be high because they are prized for their meat products (Teclemariam *et al.*, 2009). Turtles are also sometimes killed in Eritrea for their fat, which is used to produce oil that is sold for large sums of money both within and outside the country (for instance, in Yemen) for medicinal purposes (Teclemariam *et al.*, 2009).

Egg harvesting has also been reported as a threat impacting green turtles in the Islamic Republic of Iran, with eggs being used for both consumption (in some cases as an aphrodisiac) and for use in traditional medicines (Mobaraki, 2011; 2007; 2004). Turtles are also sometimes harvested for consumption as well.

Historically, in India and Sri Lanka, a large number of green turtles were found in the Gulf of Mannar, the coastal waters if the Bay of Bengal, Arabian Sea near the Laccadive Island Green turtles were harvested in the southeastern waters surrounding the Indian Peninsula near Tamil Nadu for many decades from the 1940s. Records indicate that turtle meat and parts were regularly exported from Tamil Nadu to Sri Lanka then to other countries such as the U.S., Singapore, and Belgium (Kuriyan, 1950; Fernando, 1973; Chari, 1964; and Shanmughasundarun, 1968 as cited in Agastheesapillai and Thiagarajan, 1979). Lakshadweep in India was an important center for green turtle harvesting for meat and turtle products such as the trade of oil and shell (Frazier, 1980 as cited in Tripathy *et al.*, 2006).

In the 1970s, green turtles were frequently documented in the Gulf of Mannar, India. Green turtles were caught regularly in the Gulf of Mannar and Palk bay where an average of 3,000 to 4,000 green turtles was harvested annually (Jones and Fernando, 1973). Green turtle export was banned in the 1980s; however, subsistence harvesting continues (Bhupathy and Saravanan, 2006).

An increase in the number of green turtles killed by fishers has been reported in Agatti Island, Lakshadweep, India. The cause for the killing has been linked to increases in green turtles within the area. The perception is that green turtles damage fishing gear and overgraze seagrass thereby reducing catch levels (Arthur *et al.*, 2013).

9.2.5.3. Factor C: Disease or Predation

The prevalence of FP in the North Indian DPS is not known. The best available data suggest that current nest and hatchling predation occurs on several North Indian DPS nesting beaches, although the magnitude of the threat is unknown.

Predation of hatchlings and eggs by red foxes (*Vulpes vulpes arabica*) is common at the Ras al Jinz, Oman, green turtle nesting beach (Mendonça *et al.*, 2010) and depredation by feral dogs has been identified as a major threat at sea turtle nesting beaches in Pakistan (Asrar, 1999; Firdous, 2001) and the main green turtle nesting beach at Ras Sharma (Stanton, 2008). On two Egyptian Red Sea beaches (Ras Honkorab and Om Al-Abath beaches, which are both within Wadi Gimal National Park limits), predation is reported to be very high with only a few nests surviving (Mancini, 2012). The most common predators observed on these two beaches in Egypt were desert foxes (*Vulpes zerda*) and dogs (*Canis lupus familiaris*), but ghost crabs were regularly observed near nests. 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). Along the beaches of Gujarat in India, dogs, jackals, monitor lizards, crabs, crows, and possibly hyenas and feral pigs depredate nests and eat hatchings (Sunderraj *et al.*, 2006a).

9.2.5.4. 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 partially address direct and incidental take of North Indian DPS green turtles, these regulatory mechanisms are insufficient or are not being implemented effectively to address the needs of green turtles. The analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels.

In areas such as Lakshadweep, India, the killing and poaching of green turtles have been banned since 1972, however, due to the lack of awareness and implementation, turtles continue to be harvested (Tripathy *et al.*, 2006). We find that the threat from the inadequacy of existing regulatory mechanisms impacts to nesting beach habitat (Factor A), overutilization (Factor B), and fishery bycatch (Factor E) continue throughout the DPS to varying degrees and may adversely affect this DPS.

There are several international treaties and/or regulatory mechanisms that pertain to the North Indian DPS, and nearly all countries lining the North Indian DPS have some level of national legislation directed at sea turtle protection (see Section 9.2.6 below). 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).

9.2.5.5. Factor E: Other Natural or Manmade Factors

The North Indian DPS of the green turtle is negatively affected by both natural and anthropogenic impacts. Fishery bycatch (longline, gill net, and trawl fishing) occurs throughout the DPS and affects juvenile to adult size turtles. In addition, pollution, vessel strikes, 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

Sea turtle bycatch from gill nets, trawls, and longline fisheries is a significant cause of sea turtle mortality in the North Indian DPS although there is less bycatch data than for other regions of the world (Wright and Mohanty, 2002; Project GloBAL, 2007; Bourjea *et al.*, 2008; Abdulqader, 2010; Wallace *et al.*, 2010b). The magnitude of trawl, gill net, and longline fisheries within the North Indian DPS is great with no substantive sea turtle protection measures in place to reduce sea turtle bycatch mortality. Along the coast of Ra's Al Hadd, one of the densest nesting beaches in this DPS, fishery related mortality is particularly high where green turtles are incidentally caught in fishing gear (Salm, 1991). The number of fishing vessels in Gujarat, India, has increased 14 percent from 1977 to 1992, likely to cause increased green turtle mortality (Sunderraj *et al.*, 2006a).

Longline Fisheries

In 2012, the Indian Ocean Tuna Commission began requiring its 31 Contracting Parties to report sea turtle bycatch and to use safe handling and release techniques for sea turtles on longline vessels (Hykle, 2013). Only 23 parties filed reports for 2012, and many were incomplete so little actual bycatch data were useful. Based on this incomplete reporting, however, we can ascertain the size of the longline fleet in the Indian Ocean. There are more than 30,000 longline vessels fishing in the Indian Ocean.

Gill Net Fisheries

Gill nets are widely deployed and used throughout the region and known to kill thousands of sea turtles in some regions (Project GloBAL, 2007). Two member Indian Ocean Tuna Commission parties, Iran and Kenya, alone reported the use of 12,023 gill nets in the Indian Ocean in 2012. In Lakshadweep, India, the most common fisheries include gill net, shore seine, anchor net and drag nets, and pole and line known to incidentally catch green turtles (Tripathy *et al.*, 2006). Along the coast of Tamil Nadu, gill nets are the most common fishing gear used with 74 green turtles reported killed during November 2001 to April 2001 (Bhupathy and Saravanan, 2006).

A bycatch survey administered off the northwestern, western, and southwestern coasts of Sri Lanka between September 1999 and November 2000 reported 5,241 total turtle entanglements, of which 908 were green turtles (Kapurusinghe and Cooray, 2002; Kapurusinghe *et al.*, 2005). Although these interactions were reported for a variety of different gear types, gill nets were identified as the key fishing gear responsible for turtle bycatch in Sri Lanka.

In Somalia, gill nets used for shark fishing have been reported as a major source of sea turtle mortality (Nurarale, pers. comm as cited in Bourjea *et al.*, 2008).

In Eritrea, gill nets are one of the most widely used gear types used by local fishermen, and fishermen reported that wide-mesh nets and many kinds of gill nets entangle turtles (Teclemariam *et al.*, 2009). Teclemariam *et al.* (2009) reported that most cases of sea turtle slaughtered within the southern Red Sea region are from turtles captured as a result of entanglement in gill nets and drift nets. They further reported that it is very common to see carapaces of killed green and hawksbill turtles along the coastline of the southern Eritrean Red Sea region.

Incidental capture of sea turtles in fishing nets (presumably in gill nets or set nets) has been identified as the main cause of mortality of juvenile green turtles within Iranian foraging areas (Mobaraki, 2007).

In the United Arab Emirates, drowning of sea turtles in abandoned fishing nets in and around the seagrass and reef habitats has been identified as a major cause of mortality (Al-Abdessalaam *et al.*, 2008).

In Qatar, entrapment of turtles in fishing nets has been identified as one of the most common threats (Tayab and Quiton, 2003). Al-Muraikhi *et al.* (2005) also identified entrapment of sea turtles in artisanal and commercial fishing operations in Qatar as a key source of mortality. Although Al-Muraikhi *et al.* (2005) did not identify the gear types responsible, fishing methods used in Qatar include gill net, large wire traps, small gargoor, and hook-and-line (Food and Agriculture Organization of the United Nations, 2001).

Trawl Fisheries

Shrimp trawling occurs in many countries throughout the North Indian DPS including Pakistan, India, Bahrain, and Saudi Arabia. Pakistan requires the use of TEDs to meet U.S. Section 609 requirements for exporting shrimp to the U.S., but inspections have not been able to confirm compliance in recent years (E. Possardt, USFWS, pers. obs., 2013). India requires the use of TEDs in shrimp trawls but compliance is unclear. Nowhere else in the North Indian DPS are TEDs being used and it can be assumed that significant sea turtle bycatch occurs. One documented assessment of the impact of trawling on sea turtles in this region is from Bahrain where trawls were reported to capture over 300 sea turtles annually, mostly greens (Abdulqader, 2010; Abdulqader and Miller, 2012); however, the level of mortality is unknown. A sea turtle bycatch assessment of trawls in Oman estimated 581 sea turtles (species unidentified) were caught by eight trawlers over a period of one year with an estimated mortality of 50 percent (Hare, 1991).

In Yemen, trawling is believed to be a significant threat to sea turtles, mainly hawksbill and greens; however, no data are available (Bourjea *et al.*, 2008).

In Eritrea, 3,342 sea turtles, 1,819 of which were green turtles, were reported as being incidentally caught by industrial shrimp and fish trawlers between 1994 and 2004 on different fishing grounds within the Eritrean Red Sea (Teclemariam *et al.*, 2009). The mortality rate of all species combined was 22 percent. Howe *et al.* (2004) identified incidental capture in nearshore trawls as the suspected cause of a mass mortality event (over 250 dead green turtles) that occurred in Eritrea in the spring of 2003.

Other Gear Types

In 2012, the IOTC began requiring its 31 Contracting Parties to report sea turtle bycatch (Hykle, 2013). Only 23 parties filed reports for 2012 and many were incomplete so little actual useful bycatch data was useful. Based on this incomplete reporting, over 70 industrial purse seiners fish in the Indian Ocean.

Vessel Strikes

Boat strikes have been identified as a major cause of sea turtle mortality in the United Arab Emirates (Al-Abdessalaam *et al.*, 2008) and Qatar (Al-Muraikhi *et al.*, 2005). Boat strikes of sea turtles also have been identified as a regular occurrence in Iran and seem to be increasing in some areas (Mobaraki, 2011). Boat strikes are undoubtedly a regular occurrence throughout the Arabian Gulf and other important green turtle foraging grounds in the North Indian DPS and, cumulatively, are likely significant, but quantification is lacking.

Beach driving

Beach driving by fisherman who haul and launch boats from Ras al Jinz beach in Oman is highly problematic, and hatchling turtles are likely being struck or run over. However, no assessment has been conducted to determine the extent of impacts on nesting turtles and hatchlings (E. Possardt, USFWS, pers. comm., 2013).

Pollution

Pollution has been identified as a main threat to sea turtles in Iran (Mobaraki, 2007) and Pakistan (Firdous, 2001); however, no specific information about the type of pollution was provided. In Sri Lanka, Kapurusinghe (2006) stated that polluted inland water flows into Beira Lake and subsequently the sea, and that garbage, including polythene and plastics, dumped on beaches in some areas is washed into the sea, where they can be lethal to sea turtles. In Gujarat, India, the increase in ports and shipping traffic results in problems from oil spills, garbage, and other pollutants such as fertilizers and cement (Surderraj *et al*., 2006).

Climate Change

Similar to other areas of the world, climate change and sea level rise have the potential to affect green turtles in the North Indian DPS. A significant rise in sea level would reduce green turtle nesting habitat in the North Indian DPS. Over the long term, North Indian DPS turtle populations could also be threatened by the alteration of thermal sand characteristics from global warming (Hawkes *et al*., 2009; Poloczanska *et al*., 2009). Sand temperatures prevailing during the middle third of the incubation period determine the sex of hatchling sea turtles (Mrosovsky and Yntema, 1980). Incubation temperatures near the upper end of the tolerable range produce only female hatchlings while incubation temperatures near the lower end of the tolerable range produce only male. As temperatures increase, there is also concern that incubation temperatures will reach levels that exceed the thermal tolerance for embryonic development, thus increasing embryo and hatchling mortality.

Thus, climate change impacts could have profound long-term impacts on green nesting in the North Indian DPS, but it is not possible to project the impacts at this point in time.

Natural Disasters

Natural environmental events also affect green turtles in the North Indian DPS. Cyclones occasionally develop in the North Indian Ocean and can disrupt green turtle nesting activity and hatchling production, but the results are generally localized and rarely result in whole-scale losses over multiple nesting seasons. Cyclones occurring in consecutive years in 1998 and 1999 in Kachchch, India, caused serve erosion of the nesting beach (Surderraj *et al*., 2006). However, when combined with the effects of sea level rise, there may be increased cumulative impacts from future storms.

9.2.6. Summary of Existing Conservation Efforts

While conservation efforts for the North Indian DPS are extensive and expanding, they still remain inadequate to ensure the long-term viability of the population. Efforts have been largely focused on the nesting beaches and there are only recent efforts underway to understand the extent of green turtle interactions with gill nets and trawlers and the resulting cumulative effects from bycatch—one of the major threats to this DPS. Concerted efforts to identify and protected critical foraging grounds is also lacking. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

9.2.6.1. Local and National Legislation and Protection

Most North Indian DPS countries have laws to protect sea turtles and/or nesting habitats. They are summarized below. The overall effectiveness and enforcement of these laws varies among the countries.

Bahrain

Ministerial Order No. 2 was issued in 2002 and prohibited the capture of sea turtles, seals, sea cows, or any other sea mammals or the abuse of their habitats along the coastline of the Kingdom or within its territorial waters. In addition, Legislative Decree No. 21 of 1996 with respect to the Environment requires the adoption of plans and policies to prevent the deterioration of the environment, which includes the requirement for mitigation measures on projects that have the potential to impact the environment (Naser, 2012).

Djibouti

Djibouti Decree 80-62/PR/MCTT of 1980 provides for protection of the sea bottom and marine fauna. This decree prohibits the capture of marine mammals and sea turtles, trade in or export of these animals, the collection of turtle eggs, and spearfishing (Hariri *et al.*, 2000). However, according to Witsen (2012), Djibouti's government has only recently begun to more actively participate in sustainable environmental practices and conservation.

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).

Eritrea

Eritrean Fisheries Proclamation No. 104 of 1998 prohibits the direct harvest of and domestic trade in protected species, including sea turtles, their eggs, and sea turtle parts and products, and protects important sea turtle habitats (Teclemariam *et al.*, 2009). According to Article 12 of the Fisheries Proclamation: (1) No person shall fish for any marine mammal or other protected species in Eritrean waters, and (2) Any marine mammals or other protected species caught accidentally shall be released immediately and returned with the least possible injury to the waters from which it was taken, whether dead or alive. Also, according to Legal Notice No. 39 of 1998 under Protected Species, Article 11 of the Proclamation, "All species of marine turtles are protected species for the purposes of Article 12 of the Proclamation and accordingly fishing for marine turtles in Eritrean waters is prohibited."

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 addition, the Forest Department in Gujarat and the Prakruthi Nature Club, are actively involved with nesting beach conservation including maintaining hatcheries, education and outreach, and rescue of stranded turtles.

Iran

In 1971, all sea turtle species were declared to be protected species by the Iranian Department of the Environment (Firouz, 2005).

Iraq

We are not aware of any national laws specifically protecting sea turtles in Iraq.

Kuwait

Kuwaiti law prohibits human consumption of sea turtle meat and eggs, and prohibits harm to sea turtles (Alhafez, 2010).

Oman

The following laws provide legal protection for sea turtles in Oman: Royal Decree 53/81, Law of Sea Fishing and Protection of Marine Biological Wealth; Royal Decree 114/2001, Law on Conservation of the Environment and Prevention of Pollution; and Ministerial Decisions MD 207/93, Forbidden to Hunt, Trap or Shoot Animals or Birds in the Sultanate.

The important nesting beaches adjacent to the Town of Ras al Hadd, Oman, are protected. However, they do not have an adequate buffer or topography to protect the nesting turtles and hatchlings from lights behind the beach (E. Possardt, USFWS, pers.comm., 2013). Although a wall approximately 400 meters long and 1 meter high has been constructed to deter females from wandering away from the nesting beach towards the lights after nesting and to deter hatchling misorientation and disorientation (R. Baldwin, Five Oceans LLC, pers. comm., 2013), it is an incomplete solution to the nesting female and hatchling misorientation and disorientation problem.

In addition to providing protection, the Oman Ministry of Environment and Climate Affairs has been conducting a tagging program at Ras al Hadd Nature Reserve for over 30 years. In 2009, the Ministry of Tourism also completed construction of a Scientific and Education Center behind Ras al Jinz Beach to support its turtle ecotourism program and conservation program.

In 2008, a Government Decree in Oman was issued prohibiting the use of bottom trawlers for benthic fishing (http://faolex.fao.org/cgi-bin/faolex.exe?rec_id=082541&database=FAOLEX&search_type=link&table=result&lang=eng&format_name=@ERALL).

Pakistan

The Sindh Wildlife Protection Act (1993) and Sindh Wildlife Protection Ordinance (1972) provide legal protection for sea turtles in Pakistan (Firdous, 2001).

During the 1980 and '90s, active and extensive conservation programs were being implemented on the important nesting beaches by Sindh Wildlife Department and World Wildlife Fund in Pakistan. These included beach monitoring, tagging, and protection of eggs through extensive use of hatcheries.

Qatar

Qatari Law No (5) on endangered wildlife and their products prohibits the capture of green turtles and the collection of turtle eggs.

Saudi Arabia

In the Kingdom of Saudi Arabia marine turtle protection is governed by two laws providing the necessary legislative frameworks:

(1) The Fishing Law – Hunting, exploitation, and protection of the marine living natural resources in the territorial waters of the Kingdom of Saudi Arabia is regulated by Ministerial Decision number 21911 dated on 27/3/1409H equivalent to 6/11/1988G issued by the Minister of Agriculture defining the Executive Bill of the law issued by the Royal Decree number M/9 dated 27/3/1408H equivalent to 18/11/1987G. In addition to regulating all fishing and maritime commercial exploitation, this law *inter alia*, prohibits the taking of marine mammals, marine turtle and seabird eggs. Competent authorities involved in implementation of this law in Saudi Arabia are the Ministry of Agriculture and Water, the Ministry of Interior, and the National Commission for Wildlife Conservation and Development.

(2) Marine Scientific Research Regulation – This law regulates all research in territorial waters of Saudi Arabia, which includes all technical and scientific activities conducted in marine areas including recording, aquatic studies and research as well as marine treasures in the territorial waters of the Kingdom of Saudi Arabia. This law was issued by a Ministerial Decision number 103 dated on 10/8/1413H equivalent to 1/2/1993G, approved by the Royal Decree number M/12 dated 11/8/1413H equivalent to 2/2/1993G. The competent authority empowered with the implementation of this law in Saudi Arabia is the Department of Military Survey, The Ministry of Defense and Aviation.

Somalia

There do not appear to be any national laws specifically protecting sea turtles in Somalia. In addition, national environmental legislation is insufficient for meaningful management and protection of habitats and resources, and since the outbreak of civil war in 1991, existing national laws and regulations have not been enforced (PERSGA, 2001).

Sri Lanka

A 1993 amendment to the Fauna and Flora Protection Ordinance of 1972 strengthened legal protection of sea turtles in Sri Lanka and now completely outlaws the killing, injuring, or keeping of sea turtles or their eggs in captivity. In addition, the main green turtle nesting beach in Sri Lanka lies within the Rekawa Protected Area.

Since 1996, The Turtle Conservation Program – Sri Lanka has been conducting surveys and patrols to protect the green turtle population at Rekawa Protected Area, one of the two most important green turtle nesting sites in Sri Lanka.

Sudan

There do not appear to be any national laws specifically protecting sea turtles in Sudan. The main law governing resource exploitation and environment is the Marine Fisheries Ordinance of 1937, which was amended in 1975 and 1978 (Hariri *et al.*, 2000). The Ordinance provides for the establishment of closed or restricted fishing areas and fisheries licensing; prohibits collection of shells, aquarium fishes, and coral; and prescribes minimum allowable sizes for fish species and allowable methods of fishing. The extent to which this ordinance may provide protection to sea turtles is unknown.

United Arab Emirates

Federal Law 23 of 1999 pertaining to the Exploitation, Protection and Development of Living Aquatic Resources, and Federal Law 24 of 1999 pertaining to the Protection and Development of the Environment accord full protection to sea turtles in the United Arab Emirates. These laws, as well as their bylaws and other regulations, prohibit the commercial exploitation and hunting of sea turtles throughout the coastal waters of the United Arab Emirates (Al-Abdessalaam *et al.*, 2008).

Yemen

Law 49 of 1991 prohibits the killing of sea turtles throughout Yemen; however, a well-defined framework for enforcement of this law is lacking (Hariri *et al.*, 2000).

Two Yemeni NGOs, the Environment Friends Society established in 2000, and Halfoon Wildlife Protection Society established in 2012, have been patrolling and surveying some parts of Sharma Protected Area. The Yemeni Biological Society, with funding support from USFWS, is planning to work with these groups to establish index beach surveys along at least 10 km of nesting beach to determine nest and hatching success, and to work with local communities to protect nesting turtles on Sharma Protected Area in 2014.

9.2.6.2. International Instruments

In 2012, the Indian Ocean Tuna Commission (IOTC) began requiring its 31 contracting Parties to report sea turtle bycatch and to use safe handling and release techniques for sea turtles on longline vessels. The IOTC and IOSEA also recently completed an "Ecological Risk Assessment and Productivity -Susceptibility Analysis of sea turtles overlapping with fisheries in the IOTC region." One conclusion was that green turtles account for 50-88 percent of artisanal and commercial gill nets bycatch. Two methods of estimating total bycatch were used, and resulted in an annual gill net bycatch estimate of 29,488 sea turtles within the IOTC region.

Several regulatory mechanisms that apply to green turtles regionally or globally apply to green turtles within the North 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
- Fishery and Agricultural Organization Technical Consultation on Sea Turtle-Fishery Interactions
- Indian Ocean Southeast Asian Marine Turtle Memorandum of Understanding (IOSEA)
- Indian Ocean Tuna Commission
- International Convention for the Prevention of Pollution from Ships (MARPOL)
- International Union for Conservation of Nature
- Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region
- Ramsar Convention on Wetlands
- United Nations Convention on the Law of the Sea
- United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing
- United States Magnuson-Stevens Fishery Conservation and Management Act

9.3. 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).

Almost 75 percent of the nesting occurs in a small geographic portion of the DPS (Yemen and Oman). There is also some substantial nesting in Pakistan, Saudi Arabia, and India and along the

Red Sea. Threats are generally uniform throughout the DPS, and there is limited data on population trends. There is no evidence that there is any one nesting site at more risk than others. Therefore, 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.

9.4. Assessment of Extinction Risk

For the SRT's assessment of extinction risk for green turtles in the North 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 elements considered for this region (Table 9.3), 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 9.4). See section 3.3. for details on the six elements and the voting process.

Table 9.3. Summary of ranks that reflect the importance placed by each SRT on the critical assessment elements considered for the North Indian DPS. For Elements 1-4, higher ranks indicate higher risk factors.

	Critical Assessment Elements					
	Element 1	Element 2	Element 3	Element 4	Element 5	Element 6
	Abundance (1 to 5)	Trends / Productivity (1 to 5)	Spatial Structure (1 to 5)	Diversity / Resilience (1 to 5)	Five-Factor Analyses (–2 to 0)	Conservation Efforts (0 to 2)
MEAN RANK	1.4	2.0	1.6	2.0	-0.9	0.3
SEM	0.2	0.2	0.2	0.2	0.2	0.1
RANGE	1–2	1–3	1–2	1–3	(-2)–0	0–1

With respect to the important rankings for the six critical assessment elements, nesting abundance featured most prominently in the risk threshold voting, likely owing to the overall large population size in the North Indian DPS. Spatial structure (i.e., limited overall nesting distribution) featured relatively high in the risk threshold voting. SRT members also generally thought that future threats not yet reflected in nester abundance or not 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) is for the Diversity / Resilience and Trends / Productivity Sections (Table 9.3).

Table 9.4. Summary of Green Turtle SRT member expert opinion about the probability that the North Indian DPS will reach quasi-extinction under current management regimes, 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.

	Probability Of Reaching Quasi-Extinction					
	<1%	**1-5%**	**6-10%**	**11-20%**	**21-50%**	**>50%**
MEAN ASSIGNED POINTS	68.4	19.8	9.7	1.7	0.5	0.0
SEM	7.0	4.2	3.2	0.7	0.4	0.0
Min	20	2	0	0	0	0
Max	98	50	40	5	5	0

Of the six categories describing the probability that the North Indian DPS will reach quasi-extinction within 100 years (Table 9.4), the SRT member votes resulted in the greatest point designations (i.e. probability) in the '<1%' and '1-5%' risk ranges (mean of 68.4 and 19.8points, respectively). The '>50%' and '21-50%' ranges received the fewest points from SRT members (mean of 0 and 0.5, respectively).

In their vote justifications, most members cited the high, relatively stable, nesting numbers on protected beaches within one region weighed against the current threats of coastal development and fishery practices. Additional factors that were cited included the uncertainty in abundance, political instability, and limited government focus on conservation. In general, the vote justifications provided for this DPS were relatively consistent across SRT members.

9.5. Synthesis and Integration

During the analysis of the status of the North Indian DPS, an integrated approach was taken by the SRT to consider the many critical elements described earlier. 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. The North Indian DPS has expansive, largely undeveloped nesting beaches and many of the most important green turtle beaches are protected from development as nationally designated reserves or protected areas, although threats still remain. The North Indian DPS also features vast coastal sea grass beds distributed throughout the region, which provide abundant foraging grounds for this species. There is little if any known interchange with the larger Indian Ocean to the south, although this may be an artifact of the paucity of genetic data, flipper tag returns and satellite tracking studies.

Population trends and spatial structure in the North Indian DPS were considered by the SRT to have low likelihood of contributing to the extinction of the DPS in the next 100 years, and the other two population factors were also rated relatively low with regard to extinction risk. Large robust nesting populations in Yemen and Oman and a broad distribution of smaller and healthy nesting populations dispersed throughout the region.

Coastal development, beachfront lighting, fishing practices, and marine pollution at nesting beaches and important foraging grounds are continuing concerns across the DPS. Current illegal harvest of green turtles and eggs for human consumption is a continuing but limited threat to this DPS. Fishery bycatch occurs throughout the North Indian DPS, particularly bycatch mortality of green turtles from gill nets and trawl fisheries, and the cumulative mortality from these fisheries is probably the greatest threat to this DPS. Additional threats from boat strikes, which are becoming more common, and result of climate change will negatively affect this DPS. The SRT considered all of these threats in the overall extinction risk analysis.

Conservation efforts are substantial but uneven in the North Indian DPS and focused almost entirely on nesting beaches. The ability for some countries to sustain or develop needed conservation programs in the context of political instability within the region is of concern. Further, our analysis did not consider the scenario in which current laws or regulatory mechanisms were not continued. Given the conservation dependence of the species, without mechanisms in place to continue conservation efforts in this DPS, some threats could increase and population trends could be affected.

SRT members attributed the largest probability (67.4) to the lowest single category of extinction risk (<1%), with a probability of 88.2 that reaching quasi-extinction is <5 percent, and a probability of 97.9 that reaching quasi-extinction is <10 percent. These results reflect fairly low risks in many of the critical elements, most notably in abundance, although concerns remain about the level of threats from some factors and the inadequate level of conservation measures in place, especially in addressing cumulative fisheries bycatch impacts in the DPS.

10. EAST INDIAN-WEST PACIFIC DPS (DPS #6)

10.1. DPS Range and Nesting Distribution

The western boundary for the East Indian–West Pacific DPS is 84°E longitude from 40°S to where it coincides with India near Odisha, northeast and into the West Pacific Ocean to include Taiwan extending east at 41°N to 146°E longitude, south west to 4.5°N, 129°E, then south and east to West Papua in Indonesia (at 135°E) and the Torres Straits in Australia (at 142°E longitude). The southern boundary is 40°S latitude, 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 5,000 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.

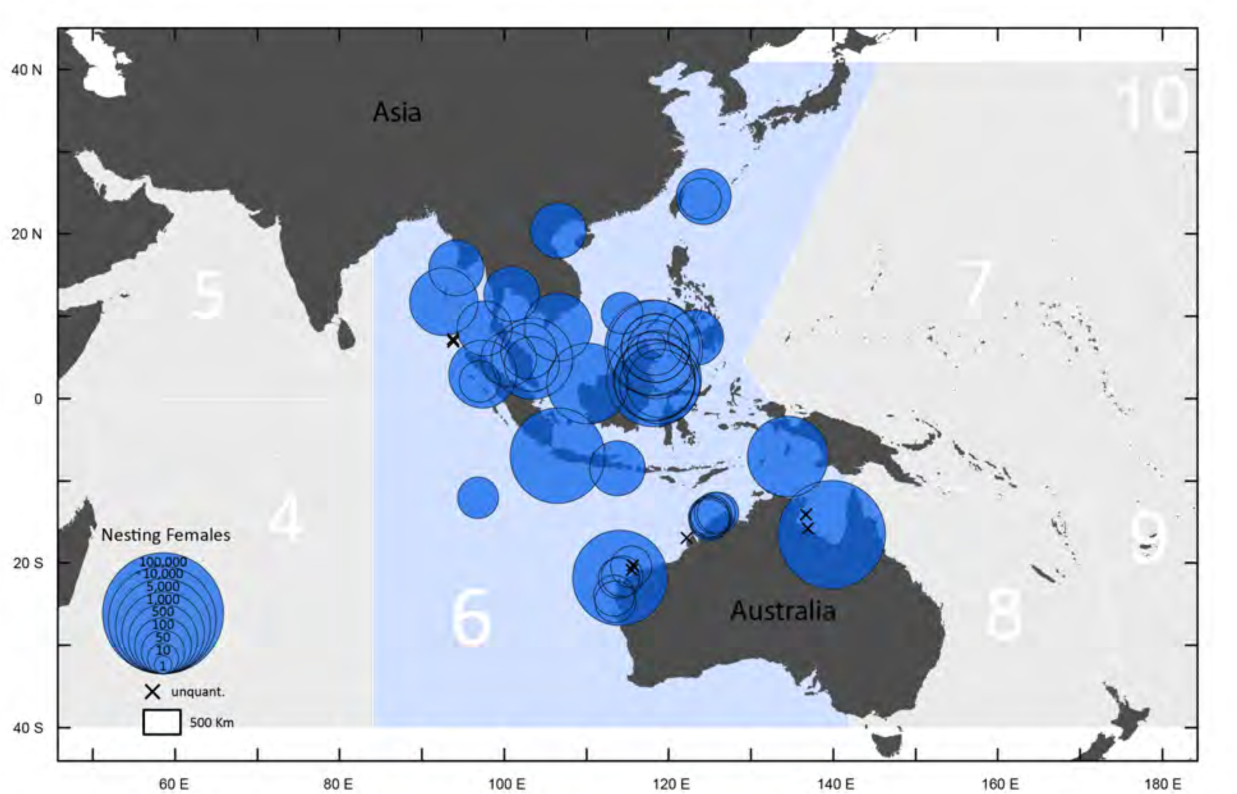

Figure 10.1. Nesting distribution of green turtles in East Indian-West Pacific DPS (blue-shaded area). Size of circles indicates estimated nester abundance (see Section 10.2.1). Locations marked with '×' indicate nesting sites lacking abundance information.

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 Kimberly region of Australia, the green turtle appears to have a broad 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.*, 2006a).

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 Baguan 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 [(Total Counted Females/Years of Monitoring) x 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.

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Australia	Ashmore Reef	1994 (3 weeks) and 1998 (2 weeks)	1
Australia	Barrow Island	1998–2004	Not calculated (974 crawls observed)
Australia	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	1999	900
Australia	Cartier Island	1998	Not calculated (1 nest observed)
Australia	Cape Range NP	2008–2010 (40 days of monitoring)	30
Australia	Coral Bay	2008–2009 (34 days of monitoring)	7
Australia	Lacepedes Islands	2006	Not calculated (500–1,000 crawls observed)
Australia	Lowendal Island group	1998–2003	Not calculated (4 crawls observed)
Australia	Ningaloo, North West Cape	2009–2010	6,269
Northern Australia	Wellesley Group (3 sites of Bountiful Island, Pisonia and Rocky Islands near Mornington Island)	No survey year provided[1]	25,000

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Northern Australia	Northern Territory (Arnhem Wessel, Cobourg, Groote, Groote Eyelant, Pellew, Tiwi, Cobourg, and Cobourg Peninsula)	1991–2004	Not calculated
Brunei	Brunei	2004	1
India	Great Nicobar Island	1991	Not calculated (13 crawls observed on 10 beaches)
India	Little Nicobar Island	1991	Not calculated (1 crawl observed)
India	Andaman and Nicobar Islands, India	2001	750
Bangladesh	St. Martin Island	1996 to 2001	23
Indonesia	Amandangan	2009	905
Indonesia	Bangkaru	1999	62
Indonesia	Belambangan Island	2000	1,736
Indonesia	Bilang-Bilangan (Berau Islands)	2008–2009	7,156
Indonesia	Derawan (Berau Islands)	2002–2006	29
Indonesia	Enu	1997–1998	2,048
Indonesia	Mataha	2008	1,652
Indonesia	Pangumbahan	2010	5,199
Indonesia	Sambit	1998–2000	555
Indonesia	Sangalaki	2003–2009	2,740
Indonesia	Meru Betiri National Park	1991–1995	296
Japan	Ibaruma Beach	1995–2003	181
Japan	Lejima	1995–1996	1
Japan	Ishigaki Island	1995–2003	56
Malaysia	Lankayan Island	1999–2001	43
Malaysia	Mentawak	2011	11
Malaysia	Pahang	2002	188
Malaysia	Perak	No survey provided[2]	150
Malaysia	Redang Island	2004–2008	278
Malaysia	Sabah Turtle Island Park (Gulissaan Island, BakkunaanKechil, Selingaan Island)	2009–2011	7,011
Malaysia	Sarawak Turtle Island	1999– 2001	1,155

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Malaysia	Sipadan	2001	585
Malaysia	Terranganu	1984–2000	1,875
Myanmar	Kaingthaung Kyun	1999	1
Myanmar	Thameehla Island	2007	109
Philippines	Panikian Island	2000	354
Philippines	Baguan Island	2008-2012	5,874[3]
Philippines	Taganak (Turtle Island)	2008-2012	637[3]
Philippines	Lihiman (Turtle Island)	2008-2012	1,217[3]
Philippines	Langaan (Turtle Island)	2008-2012	808[3]
Philippines	Great Bakkungaan (Turtle Island)	2003-2007	118[3]
Taiwan, Province of China	Lanyu	2010	19
Taiwan, Province of China	LiuChiu Island	2011	23
Taiwan, Province of China	Taipin Tao	2009	67
Taiwan, Province of China	Wan-an	2002	26
Thailand	Huyong Island	2004	105
Thailand	Khram Island, Sea Turtle Conservation Center of the Royal Thai Navy	2011	297
Thailand	Tarutao National Park	2003	1
Viet Nam	Con Dao Island	2001	900
Viet Nam	Minh Chau Island	2004	300
Viet Nam	Nui Chua Nature Reserve	2001–2004	10

1 EPA Queensland Turtle Conservation Project unpublished data as Cited in Limpus(2009)
2 Cited in Liew (2002)
3 Numbers updated based on external review subsequent to SRT voting. We don't believe the revisions would appreciably change how the SRT evaluated risk for the DPS.

Table 10.2. Green turtle nester abundance distribution among nesting sites in the East Indian–West Pacific DPS.

NESTER ABUNDANCE	# NESTING SITES DPS 6
unquantified	7
1–10	7
11–50	8
51–100	3
101–500	11
501–1000	8
1001–5000	7
5001–10000	5
10001–100000	1
>100,000	0
TOTAL SITES	**57**
TOTAL ABUNDANCE	**77,009**
PERCENTAGE at LARGEST NESTING SITE	**32% (Wellesley Group, Australia)**

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 (Meru Betiri 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, Univ. of Queensland, pers. comm., 2002). At Pangumbahan (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 Thamihla Kyun, the lack of recent and/or multiple year datasets prevents an assessment of the current trends at these sites.

For western Australia, primary nesting concentrations include the North West Cape, and the islands of Lacepede 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 the 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 in the Gulf of Thailand, Vietnam, the Berau Islands, and perhaps Enu Island, although updated information is needed for these sites. At Sipadan, Sarawak and Terengganu in Malaysia, nesting appears to be stable. Nesting has remained stable in the Philippine Turtle Islands and may have increased at the Sabah Turtle Islands.

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 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). However, the PVA analysis conducted for this review, shows a slight past, and thus a future, decline at Terengganu (see Figure 10.3). It should be noted, however, that data since 1927 (Banks, 1937) suggests that the current population, although stable, is dramatically reduced from historical levels.

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, Kolej Universeti Sainsdan Teknogli, Malaysia, pers. comm., 2002), suggesting 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.

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). Wan-an appears to have a decreasing trend, while Lanyu exhibits no apparent trend.

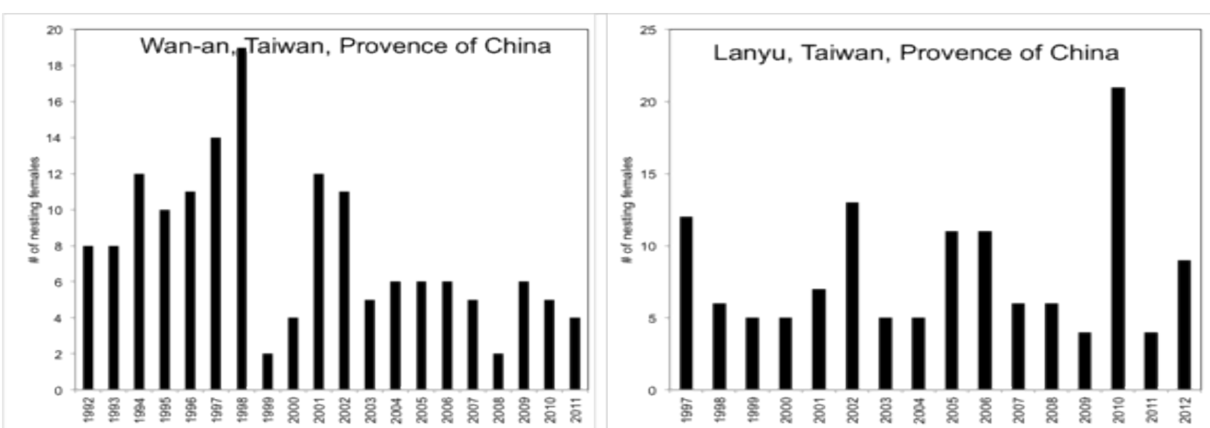

Figure 10.2. Abundance data for green turtle nesting females in the East Indian-West Pacific DPS, for which there are 10 or more years of recent monitoring data.

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 (Figure 10.3). 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. Only Sabah Turtle Islands represent a sizable nesting population, estimated at 7,011 in 2011. 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.

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.

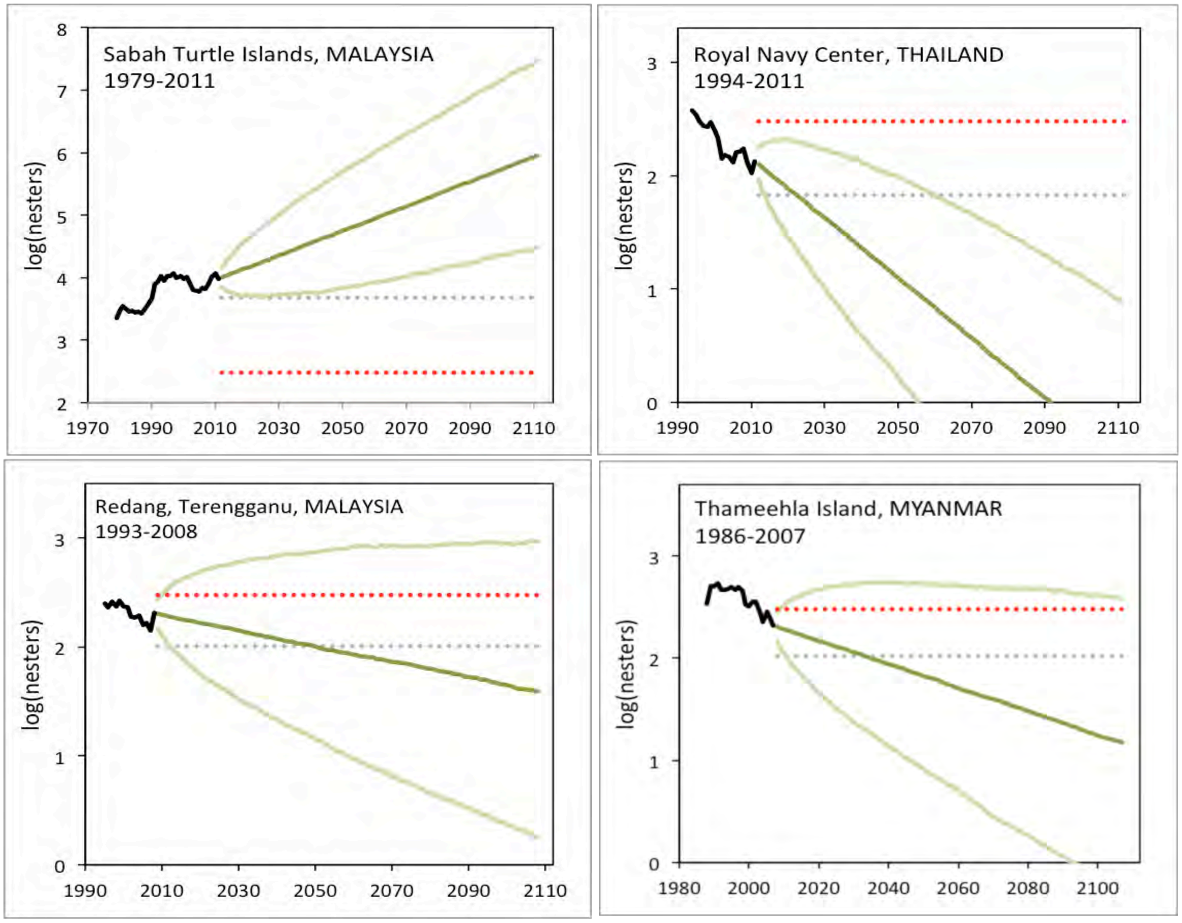

Figure 10.3. Stochastic Exponential Growth (SEG) Model Output for the Sabah Turtle Islands and Redan, Terengganu, Malaysia; the Royal Navy Center, Thailand; and Thameehla Island, Myanmar. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 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 percent decline) at the end of 100 years. There is also 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).

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). For these beaches, there is a 100 percent probability that this population will fall below the trend reference point (50 percent decline) within 100 years. There is also 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 occurred at more than 22 rookeries. There appears to be a complex population structure, even though there are gaps in sampling 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 internesting migrations, and defining the range of internesting 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 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 in the Philippines; and 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, internesting 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 82.1 cm CCL to 103.6 cm CCL (Trono, 1991; Hirth, 1997; Charuchinda and Monanunsap, 1998; Basintal, 2002). Growth rates are 0.83 cm/yr for nesting females according to 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 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. Peak nesting occurs from March to July on Derawan Island (Charuchinda and

Monanunsap, 1998; Abe *et al.*, 2003; Aureggi *et al.*, 2004; Adnyana *et al.*, 2008), and year-round in Thameela Island, Myanmar and in Aru, Indonesia, with peak nesting from November to March in Aru (Lwin, 2009a; Dethmers, 2010). Peak nesting occurs from November to March in Sukamade, southeastern Java (Arinal, 1997), Barrow Island, Australia 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.

All of these factors were considered given current management regimes. The following information on these factors /threats pertains to green turtles found in the East Indian-West Pacific DPS.

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 the East Indian-West Pacific 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 pollutionis 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.

The shoreline along the east coast of India is altered due to the construction of four major ports, which has resulted in the blockage of the littoral drift (Komar, 1983). Of the 306 islands in the

Andaman and Nicobar Islands of India, 94 are designated as wildlife sanctuaries, six of which are national parks, and two of which are marine national parks (Andrews *et al.*, 2006a). Sand 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.*, 2006a). 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 Terrengganu (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 (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 QuanLan and Minh Chau(Ministry of Fisheries, 2003), and sand mining operations on Minh Chau and QuanLanalso threaten nesting habitat.

Historically, green turtle nests were found along the coast of China from Fuijian 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. HuidongGangkou 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 topossibly 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 (Andrews, 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 (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 occurs 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 the East Indian-West Pacific 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 turtleeggs 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 Terrengganu 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, 1991a). 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, but had increased to nearly 1 million eggs by the late 1990s (E. Chan, Institute of Oceanography, Kolej Universeti Sains dan Teknogli, Malaysia, pers. comm., 2002 as cited in NMFS and USFWS, 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 (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.*, 2006a). 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 (Milliken and Tokunaga, 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 fisherman 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 presenceof 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 (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.*, 2006a). 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.*, 2006a). 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 (*Musmusculus*) 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.ioseaturtles.org/Education/seaturtlebooklet.pdf). In Japan, raccoon dogs (*Nyctereutesprocyonoides*) and weasels (*Mustelaitatsi*) 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 (*Vulpesvulpes*) 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. The analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels. 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 anthropogenicfactors 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 (Ahmed *et al.*, 2006; Khan *et al.*, 2006).

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). One of the main threats to green turtles in Vietnam is the incidental capture from gill and trawl nets and the opportunistic capture by fishers. Hundreds of green turtles are captured by fisheries per year in Vietnam (Ministry of Fisheries, 2003; Hamann *et al.*, 2006a)

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 treat 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 USFWS, 1998; Fuentes *et al.*, 2010a). 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 hatchling 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. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

Hatcheries have been set up throughout the region to protect a portion of the eggs laid and prevent complete egg harvesting. In addition, 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.

In India, since 1978, the Centre for Herpetology/ Madras Crocodile Bank Trust has conducted sea turtle surveys and studies in the islands. The Centre for Herpetology/ Madras Crocodile Bank Trust, along with the Wildlife Institute of India and Ministry of Environment and Forests, produced a series of manuals on sea turtle conservation, management and research to help forest officers, conservationists, NGOs and wildlife enthusiasts conduct sea turtle conservation and research programs (ANET, 2003 as cited in Shankar and Andrews, 2004). Recently, a consolidated manual has been produced to achieve these goals by Dakshin Foundation and Madras Crocodile Bank Trust (MCBT) (Tripathy *et al.*, 2012).

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-governmental 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 nests, 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 conducted 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 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). The level of effectiveness and progress of these goals is not known.

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 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). 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).

Illegal trade of turtle parts continues 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 to confiscate sea turtle products (MoFI, 2004 as cited in Stiles, 2009). The level of effectiveness and progress of this program is not known.

TEDs are now in use in Thailand, Malaysia, the Philippines, Indonesia and Brunei, expanded by 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 fewer 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.

As in other DPSs, persistent marine debris poses entanglement and ingestion hazards to green turtles. In 2009, Australia's Department of the Environment, Water, Heritage and the Arts published a threat abatement plan for the impacts of marine debris on vertebrate marine life (http://www.environment.gov.au/system/files/resources/d945695b-a3b9-4010-91b4-914efcdbae2f/files/marine-debris-threat-abatement-plan.pdf).

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

The Wildlife Protection Act 1978 (1984 Amendment) prohibits capturing and selling of green turtles.

Cambodia

No national legislation exists to protect sea turtles (Shanker, 2004).

China

In China, wildlife protection occurs under the Law of the People's Republic of China on the Protection of Wildlife (1988) (http://www.china.org.cn/english/environment/34349.htm), including sea turtles that are listed as state-protected Class II species. Class II species can be hunted for research, domestication, exhibition, or other special purposes if approved by Provincial agencies. Illegal trafficking and hunting of protected species is punishable with prison terms and fines (Lam *et al.*, 2011). China promulgated the Law of Wildlife Protection in 1989 and the Ordinance of Aquatic Wildlife Protection in 1993. In Guangdong Province, the Rule of Guangdong Sea Turtle Resources Protection [1988] was promulgated (Wang, 2006).

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).

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, 2002).

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 Maison *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 (TanjungJara, 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

The catching and commercial exploitation of sea turtles and their products in Vietnam was prohibited in April 2002 by Government Decree 48/2002/ND-CP and since by Circular 02/2006/TT-BTS of the Ministry of Fisheries, which supplements Government Decree 59/2005/ND-CP of May 2005 (Stiles, 2009). Circular 02/2006/TT-BTS of the Ministry of Fisheries dated March 2006 supplements Government Decree 59/2005/ND-CP of May 2005, which outlines management and controls on marine resources.

10.2.6.2. International Instruments

There are a minimum of 17 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 instrumentslisted 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
- Indian Ocean Southeast Asian Marine Turtle Memorandum of Understanding (IOSEA)
- Indian Ocean Tuna Commission
- International Convention for the Prevention of Pollution from Ships (MARPOL)
- International Union for Conservation of Nature
- Memorandum of Agreement between the Government of the Republic of the Philippines and the Government of Malaysia on the Establishment of the Turtle Island Heritage Protected Area
- Memorandum of Understanding on Association of South East Asian Nations (ASEAN) Sea Turtle Conservation and Protection
- Ramsar Convention on Wetlands
- Secretariat of the Pacific Regional Environment Programme
- United Nations Convention on the Law of the Sea
- United States Magnuson-Stevens Fishery Conservation and Management Act
- United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing

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 nesting sites 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 once 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.

	Critical Assessment Elements					
	Element 1	Element 2	Element 3	Element 4	Element 5	Element 6
	Abundance (1 to 5)	Trends / Productivity (1 to 5)	Spatial Structure (1 to 5)	Diversity / Resilience (1 to 5)	Five-Factor Analyses (–2 to 0)	Conservation Efforts (0 to 2)
MEAN RANK	1.50	2.75	1.42	1.33	–1.42	0.50
SEM	0.19	0.22	0.19	0.19	0.23	0.19
RANGE	1–3	1–4	1–2	1–2	(-2)–0	0–2

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 quasi-extinction under current management regimes, 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.

	Probability Of Reaching Quasi-Extinction					
	<1%	**1-5%**	**6-10%**	**11-20%**	**21-50%**	**>50%**
MEAN ASSIGNED POINTS	60.50	12.75	10.25	9.83	4.17	2.50
SEM	10.97	3.64	3.68	4.90	2.94	2.09
Min	0	0	0	0	0	0
Max	99	30	40	50	35	25

With respect to the overall risk of extinction, of the categories describing the probability that the DPS will reach quasi-extinction 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 rookeriesexhibiting 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.

10.4.1. 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.

	Critical Assessment Elements					
	Element 1	Element 2	Element 3	Element 4	Element 5	Element 6
	Abundance (1 to 5)	Trends / Productivity (1 to 5)	Spatial Structure (1 to 5)	Diversity / Resilience (1 to 5)	Five-Factor Analyses (–2 to 0)	Conservation Efforts (0 to 2)
MEAN RANK	1.73	2.27	1.91	1.82	–1.27	0.64
SEM	0.27	0.24	0.21	0.23	0.24	0.20
RANGE	1–4	1–4	1–3	1–3	(-2)–0	0–2

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 quasi-extinction under current management regimes 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.

	Probability of Reaching Quasi-Extinction					
	<1%	1-5%	6-10%	11-20%	21-50%	>50%
MEAN ASSIGNED POINTS	52.00	13.73	13.73	9.64	7.27	3.64
SEM	11.31	3.17	5.57	3.60	4.18	3.17
Min	0	0	0	0	0	0
Max	95	30	60	30	45	35

With respect to the risk of extinction with SPR consideration, of the categories describing the probability that the DPS will reach quasi-extinction 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%' 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 57 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 extinction risk analysis. Approximately 16.5 percent of the votes cast for were for the '>11%' likelihood of reaching quasi-extinction of extinction within 100 years, with 23 percent cast for '1–10%' likelihood, and 60.5 percent cast for '<1%' likelihood of reaching quasi-extinction. These results reflect the widespread nesting area throughout the DPS, relatively high remaining abundance of turtles, and high level of genetic diversity.

11. CENTRAL WEST PACIFIC DPS (DPS #7)

11.1. DPS Range and Nesting Distribution

The Central West Pacific DPS has as its northern boundary 41°N latitude and is bounded by 41°N, 169°E in the northeast corner, going southeast to 9°N, 175°W, then southwest to 13°S, 171°E, west and slightly north to the eastern tip of Papua New Guinea, along the northern shore of the Island of New Guinea to West Papua in Indonesia, northwest to 4.5°N, 129°E then to West Papua in Indonesia, then north to 41°N, 146°E (Figure 11.1).

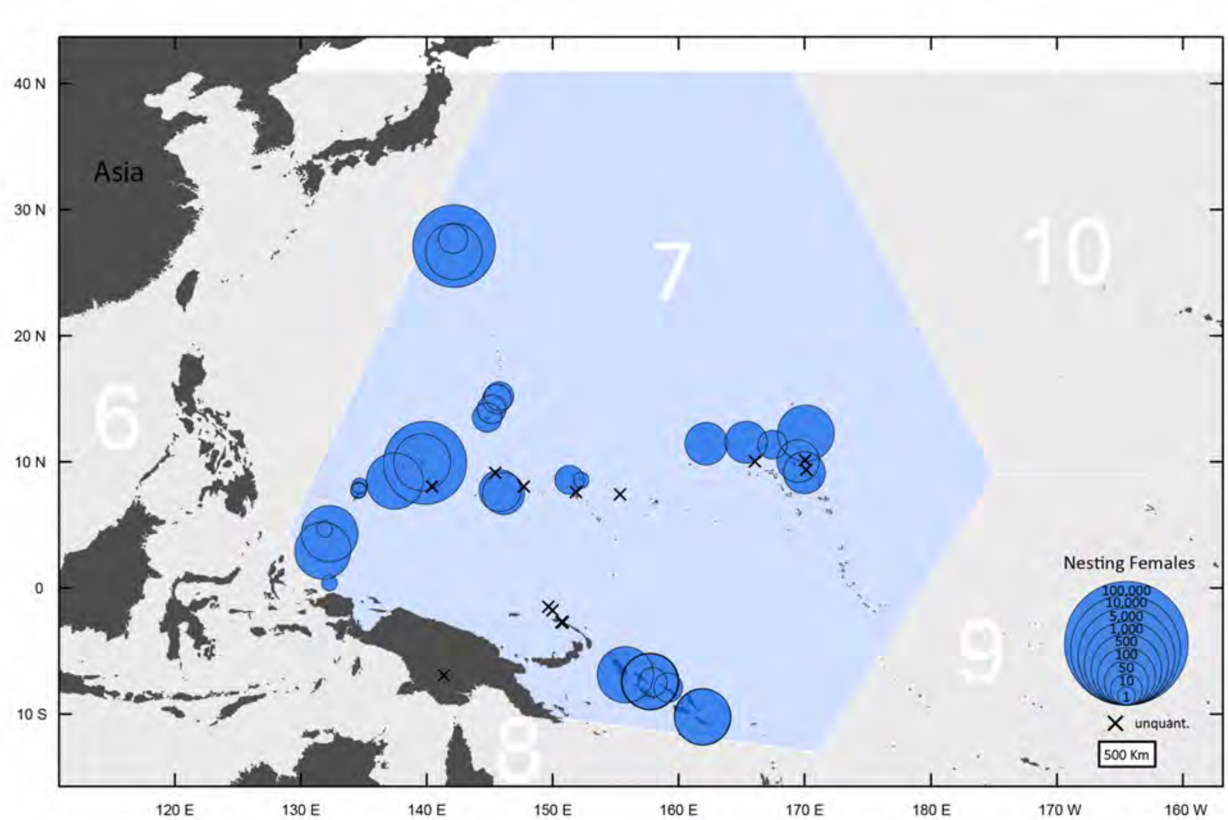

Figure 11.1. Nesting distribution of green turtles in the Central West Pacific DPS (blue-shaded region). Size of circles indicates estimate nester abundance (see Section 11.2.1). Locations marked with '×' indicate nesting sites lacking abundance information.

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, FSM (1,412); Chichijima (1,301) and Hahajima (394), Ogasawara, Japan; Bikar Atoll, Marshall Islands (300); and Merir Island, Palau (441) (NMFS and USFWS, 1998; Bureau of Marine Resources, 2005; Barr, 2006; Palau Bureau of Marine Resources, 2008; Maison *et al.*, 2010; H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012; J. Cruce, Ocean Society, pers. comm., 2013).

There are numerous other populations in the FSM, Solomon Islands, and Palau, and approximately 22 nesting green turtles in Guam, and 57 nesting 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, FSM, or remained in the Marshallese EEZ (Kabua *et al.*, 2012). Turtles tagged in Yap (FSM) 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 (H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012; Ogasawara Marine Station, Everlasting Nature of Asia. unpublished data). A turtle tagged in Japan was recorded nesting in Yap, FSM (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.*, 2005, 2006; 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 FSM, and that particular site hosts approximately 22 percent of the total annual nesting females for this DPS.

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. Sites with "n/a" have unquantified nesting.

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
CNMI	Rota	2012	15
CNMI	Tinian	2012	15
CNMI	Saipan	2012	27
FSM	Fanang	n/a	n/a
FSM	Gaferut	n/a	n/a
FSM	Oroluk Atoll	1990	n/a
FSM	Pikelot	1970	n/a
FSM	Sorol Atoll	n/a	n/a
FSM	Murilo Atoll	1993	9
FSM	Olimarao Atoll	1990	81
FSM	Elato Atoll	1993	90
FSM	East Fayu	1993	18
FSM	Ngulu Atoll	1993	192
FSM	Ulithi Atoll Loosiep Island	2010-2012	280
FSM	Ulithi Atoll Gielop and Iar Island	2010-2012	1,412
Guam	Island of Guam (and Cocos)	2010; 2008-2010	22
Indonesia	Jamursba-Medi	1995-1997	9
Japan	Mukojima	2010-2012	39
Japan	Hahajima	2010-2012	394
Japan	Chichijima	2010-2012	1,301
Marshall Islands	Ailuk	n/a	n/a
Marshall Islands	Wotho	1988	n/a
Marshall Islands	Wotje Atoll	2003	n/a
Marshall Islands	Rongerik Atoll	2003	21
Marshall Islands	Bikini	1992	75
Marshall Islands	Enewetak	1992	75
Marshall Islands	Erikub	1992	75
Marshall Islands	Jemo	1992	75
Marshall Islands	Bikar Atoll	1992	300
Palau	Pulo Ana Island	2005	1
Palau	Kayangel Atoll	2005	6

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Palau	Ngarchelong State	2005	6
Palau	Ngerechur Island	2005	6
Palau	Helen Island	2005	141
Palau	Merir Island, Sonsorol State	November 2007 to August 2008	441
Papua New Guinea	Atmago (Egmakau)	2007	n/a
Papua New Guinea	Emirau	2007	n/a
Papua New Guinea	Lemus	2007	n/a
Papua New Guinea	Mussau	2007	n/a
Papua New Guinea	Nago	2007	n/a
Papua New Guinea	Nusalaman (Nusalomon)	2007	n/a
Papua New Guinea	Ral	2007	n/a
Papua New Guinea	Usen (Usang)	2007	n/a
Solomon Islands	Kerehikapa Island	1995	32
Solomon Islands	Hakelake Island	1995	11
Solomon Islands	Ausilala	1981	225
Solomon Islands	Balaka	1981	225
Solomon Islands	Maifu	1981	225
Solomon Islands	Malaulaul	1981	225
Solomon Islands	Malaupaina	1981	225
Solomon Islands	Wagina	1981	225

Table 11.2. Green turtle nester abundance distribution among nesting sites in the Central West Pacific.

NESTER ABUNDANCE	# NESTING SITES DPS 7
unquantified	16
1 to 10	6
11-50	9
51-100	6
101-500	12
501-1000	0
1001-5000	2
5001-10000	0
>10000	0
TOTAL SITES	**51**
TOTAL ABUNDANCE	**6,518**
PERCENTAGE at LARGEST NESTING SITE	**22% (FSM)**

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 FSM (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;

Figure 11.2). 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 0 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 percent per year for the Chichijima nesting site.

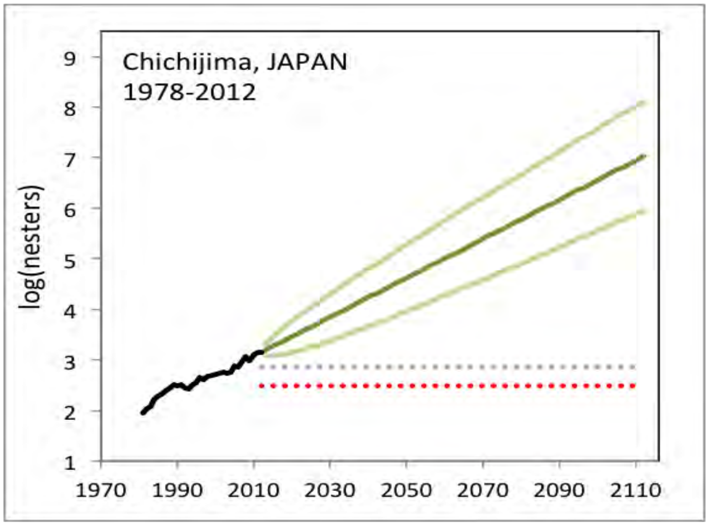

Figure 11.2. Stochastic Exponential Growth (SEG) Model Output for Chichijima, Japan. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance reference. 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 1,000 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.* (2014) suggest that there are at least seven independent stocks in the region based on mtDNA analyses.

With respect to flipper tagging, there are records of turtles tagged in the Philippines nesting in the FSM; a turtle tagged in Japan was recorded nesting in the FSM; turtles tagged in the Japan Archipelago and China were recorded nesting in the Ogasawara islands (H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012; Ogasawara Marine Center, Everlasting Nature of

Asia unpublished data); and turtles tagged in the FSM were recaptured in the Philippines, Marshall Islands, and Papua New Guinea (Palau BMR, 2008; Cruce, 2009).

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 FSM to the Philippines and to the west (G. Balazs, NMFS, unpublished data; Kolinski, *et al.,* unpublished data.)

Demographic data availability is limited and somewhat variable for nesting sites in this DPS. Variability in parameters such as remigration interval, clutch size, hatching success, 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 known mean nester sizes range from 102 cm CCL in Palau to 104.5 cm CCL in Tinian, CNMI (Pultz *et al*., 1999).

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, but occurs from November to August in Palau; from March through September in the FSM; 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.

All of these factors were considered given current management regimes. 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 FSM, 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 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 FSM 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 hatchling 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 leucocophala*) 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 FSM (NMFS and USFWS, 1998; Government of the FSM, 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 FSM: 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 FSM, 2004). 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 Palau, dredging and filling for Ollei Dock (Ngerechelong), Ngetpang 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 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, FSM, Guam, Kiribati (Gilbert Islands chain), Papua, Papua New Guinea, Republic of the Marshall Islands, and Palau (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), FSM (Cruce, 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 FSM (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, FSM, 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, 2012).

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 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 FSM, 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 FSM, nest predation by ghost crabs was reported to be a substantial problem in the 1990s on Olimarao Island, as well as a potential threat on Falipi Island, both within the Olimarao Atoll (S. Kolinski and A. Smith, pers. comm., as 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 Olimarao 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 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 Warmandi 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. The analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels. 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 FSM 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 not known to be of great consequence, except possibly in 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, 2012). 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, 2012).

Pollution

In the FSM, 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 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 hatchling production (Kasparek *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 FSM, 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 Central West Pacific DPS is protected by various international treaties and agreements as well as national and territorial laws. 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 hatchling 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. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

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, 2012b) and in the nearshore foraging areas of the FSM, 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. 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 FSM 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).

Japan, (Ogasawara Islands)

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 law in Japan that intends to conserve endangered species. It prohibits domestic assignment or transfer of endangered species listed in CITES, such as green turtles. 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.

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.

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.

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.

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 Ngerumekaol 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.

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).

United States (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.

11.2.6.2. International Instruments

A minimum of 17 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 for the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific (WCPF 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
- Memorandum of Understanding on Association of South East Asian Nations Sea Turtle Conservation and Protection
- Secretariat of the Pacific Regional Environment Programme
- Ramsar Convention on Wetlands
- United Nations Convention on the Law of the Sea
- United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing
- United States Magnuson-Stevens Fishery Conservation and Management Act

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.

	Critical Assessment Elements					
	Element 1	**Element 2**	**Element 3**	**Element 4**	**Element 5**	**Element 6**
	Abundance (1 to 5)	**Trends / Productivity (1 to 5)**	**Spatial Structure (1 to 5)**	**Diversity / Resilience (1 to 5)**	**Five-Factor Analyses (–2 to 0)**	**Conservation Efforts (0 to 2)**
MEAN RANK	2.50	2.42	2.17	2.17	–1.08	0.67
SEM	0.26	0.29	0.27	0.24	0.26	0.22
RANGE	1–4	1–4	1–4	1–3	(–2) –0	0–2

With respect to the important rankings for the six critical assessment elements, nesting abundance was featured relatively high in the risk threshold voting (with roughly 6,551 nesting females total); 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 11.4. Summary of Green Turtle SRT member expert opinion about the probability that the Central West Pacific DPS will reach quasi-extinction under current management regimes 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.

	Probability of Reaching Quasi-Extinction					
	<1%	**1–5%**	**6–10%**	**11–20%**	**21–50%**	**>50%**
MEAN ASSIGNED POINTS	43.25	19.25	13.75	12.50	8.58	2.67
SEM	10.92	5.19	3.65	4.54	4.08	1.78
Min	0	0	0	0	0	0
Max	99	60	40	40	40	20

Of the categories describing the probability that the Central West Pacific DPS will reach quasi-extinction 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 quasi-extinction 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 23.75 percent, suggesting that the extinction risk could be as high as 20 percent to over 50 percent. 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 in Guam and the CNMI, all on-the-ground conservation actions as well as financial and other resources that were afforded by the ESA, may not continue.

12. SOUTHWEST PACIFIC DPS (DPS #8)

12.1. DPS Range and Nesting Distribution

The Southwest Pacific DPS extends from the western boundary of Torres Strait (at 142°E longitude), southeast to the eastern tip of Papua New Guinea and out to the offshore coordinate of 13°S, 171°E; the eastern boundary runs from this point southeast to 40°S, 176°E; the southern boundary runs along 40°S from 142°E to 176°E; and the western boundary runs from 40°S, 142ºE north to Australian coast then follows the coast northward to Torres Strait.

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 (nGBR) 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).

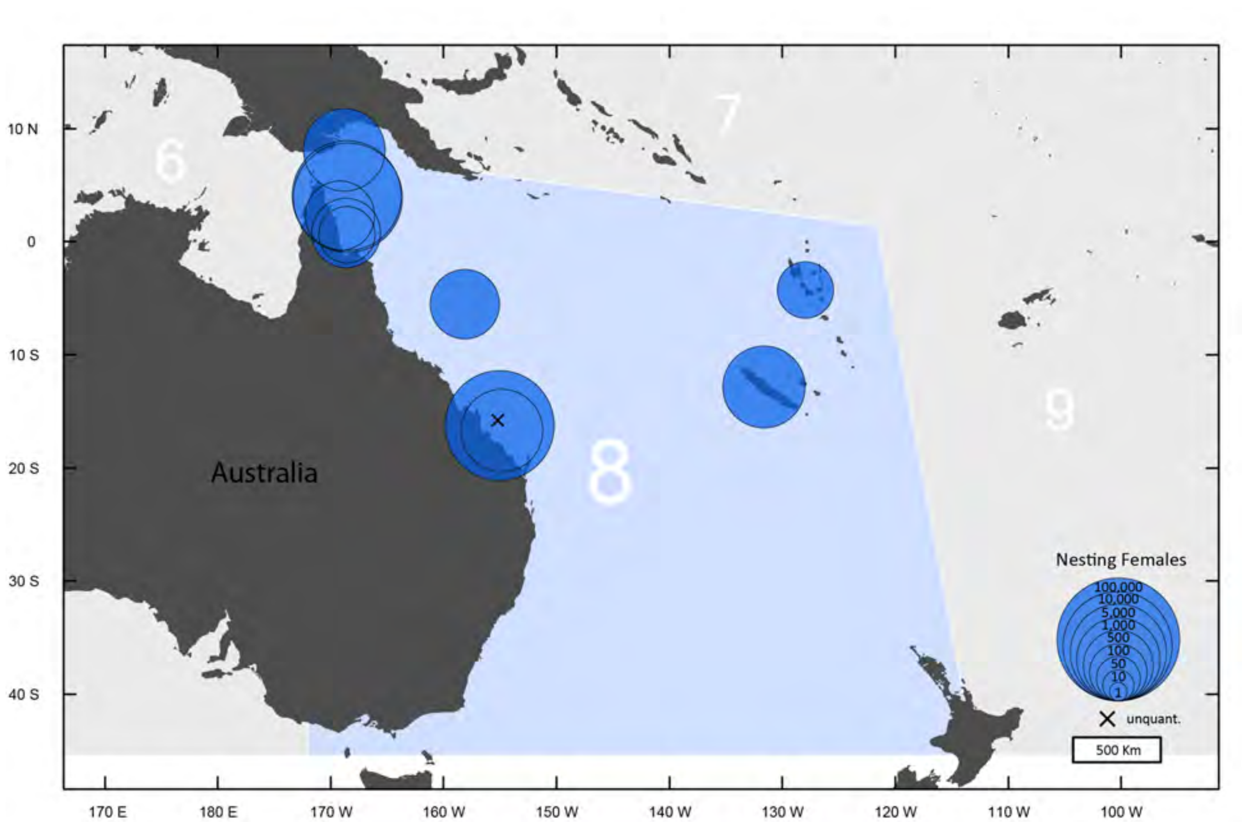

Figure 12.1. Nesting distribution of green turtles in the Southwest Pacific DPS. Size of circles indicates nesting estimated nester abundance (see Section 12.2.1). Locations marked with '×' indicate nesting sites lacking abundance information.

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 assemblages in this DPS, and perhaps the entire species (Chaloupka *et al.*, 2008), are located in the nGBR. Roughly 90 percent of the nesting activity here occurs at Raine Island and Moulter Cay, with appreciable nesting also occurring at Number Seven and Number Eight Sandbanks and Bramble Cay (Limpus, 2009). Estimates of annual nesters at Raine Island vary from 4,000 – 89,000 (Seminoff *et al.*, 2004; NMFS and USFWS, 2007; Chaloupka *et al.*, 2008; Limpus, 2009). Female nesting abundance in the nGBR is not directly counted throughout the nesting season. This is largely because of the remoteness of the site and the sheer numbers of turtles that may nest on any given night, which makes accurate counting very difficult. A mark-recapture approach (Limpus *et al.*, 2003) is used at Raine Island to estimate the number of adult female green turtles in the waters surrounding Raine Island during the sampling period. Females are painted during nightly tally counts, and then marked and unmarked adult female turtles are counted in the surrounding internesting habitats the following day using a structured survey protocol.

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), while 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 sGBR, and nearly every nesting female on Heron Island has been tagged since 1974 (Limpus and Nicholls, 2000). The mean annual nester abundance 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.

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Australia	Raine Island	2004	25,000[1]
Australia	Moulter Cay	1997–2001, 2004	15,965[2]
Australia	No. 7 Sandbank	1989, 1991,1992	> 180
Australia	No. 8 Sandbank	1997	> 637
Australia	Bramble Cay	1976, 1977, 1979, 1980	1,660
Australia	Other nGBR (including Murray Islands, other outer islands, most inner shelf cays, and mainland coast)	1981–1997	> 535[3]
Australia	Heron Island	1999–2004	4,891
Australia	Rest of Capricorn Bunker Group (Northwest, Wreck, Hoskyn, Tryon, Lady Musgrave, Masthead, Erskine, Fairfax, North Reef, and Wilson Islands)	1998/9–2003/4	31,249[4]
Australia	Rest of sGBR (primarily Bushy Island, Percy Islands, Bell Cay, Lady Elliott Island, and the mainland coast)	n/a	n/a
Coral Sea	All sites in Coral Sea	multiple ranges	1,000
New Caledonia	Huon, Leleizour, Fabre	2007–2011	1,777
Vanuatu	Bamboo Bay	2006	165[5]

[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 percent 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 percent 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 was based on external review subsequent to SRT voting, and is a significant update to the original value of 2,000. While this is a significant increase, the fact that it is a positive change suggests that SRT extinction risk estimates are conservative. With a mean ranking of 1.17 (Table 12.3), abundance was the lowest ranking element in the risk assessment for the DPS, further suggesting that this change in abundance would not have significantly affected the overall risk assessment.

[5] Based on estimates in MacKay and Petro (2013).

Table 12.2. Green turtle nester abundance distribution among nesting sites in the Southwest Pacific Ocean DPS. Each row of Table 12.1, many of which reflect aggregated nester counts within a geographic subregion, is represented as a single nesting site in this table.

NESTER ABUNDANCE	# NESTING SITES DPS 1
unquantified	1
1–10	0
11–50	0
51–100	0
101–500	2
501–1000	3
1001–5000	3
5001–10000	0
10001–100000	3
>100,000	0
TOTAL SITES	12
TOTAL ABUNDANCE	83,058
PERCENTAGE at LARGEST NESTING SITE	38% (nGBR, Australia)

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.*, 2008).

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.*, 2008). 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 percent 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.* (2008). Caution must be used when interpreting these results because they only represent females observed during one sampling bout on one night verses 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 percent 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.

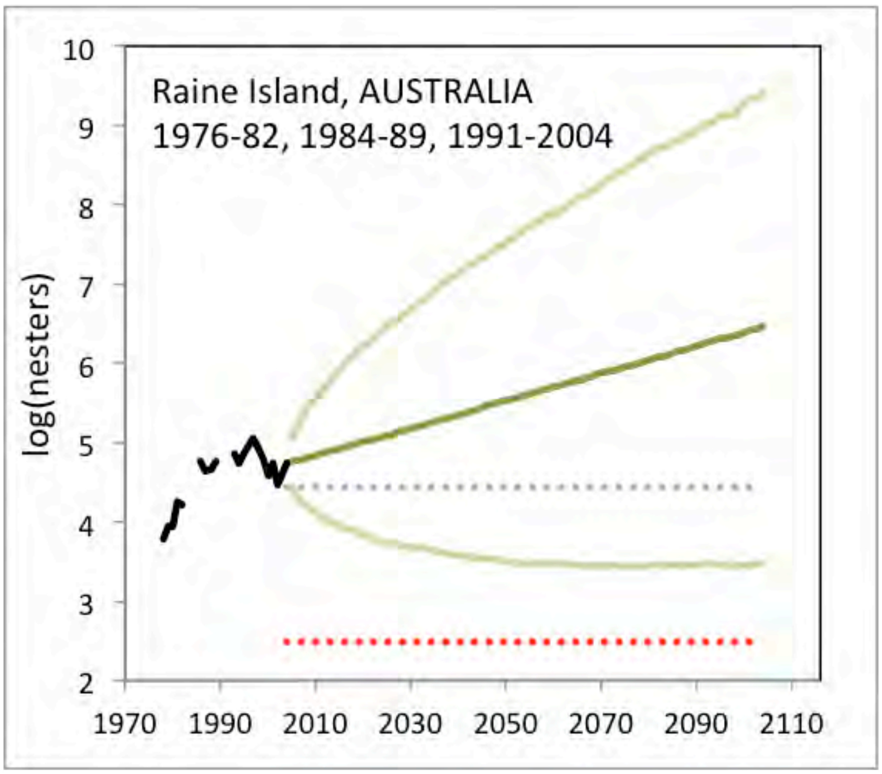

Figure 12.2. Stochastic Exponential Growth (SEG) Model Output for Raine Island, Australia. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, gray-green dotted line is trend reference, and red dotted line is absolute abundance 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.*, 2008; 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.

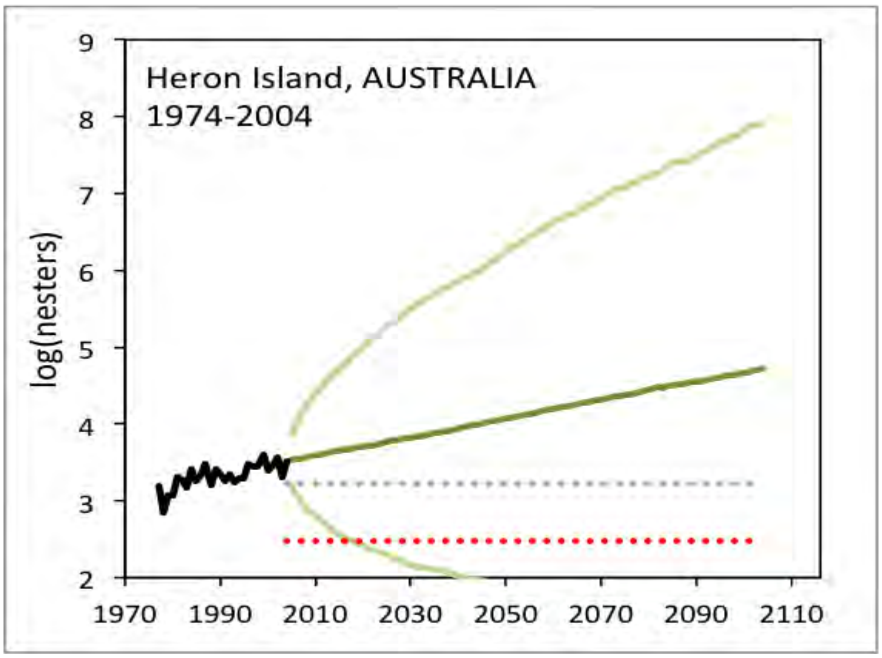

Figure 12.3. Stochastic Exponential Growth (SEG) Model Output for Heron Island, Australia. Black line is observed data, dark green line is the average of 10,000 simulations, green lines are the 2.5th and 97.5th percentiles, gray-green dotted line is trend reference, and red dotted line is absolute abundance reference.

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 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 hatchling 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.*, 2007). 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 GBR, 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; nGBR, s GBR, 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 GBR and Torres Strait regions (Jensen, 2010), but with the vast majority originating from nesting sites within the GBR. 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 stocks, 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 nGBR 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 sGBR 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 sGBR subpopulation attains maturity at 30–40 years (Limpus and Chaloupka, 1997; Chaloupka *et al.*, 2004). No similar studies are available for other regions in this population. 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, there is more than one major nesting site, 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 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.

All of these factors were considered given current management regimes. 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). Hatchling 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 hatchling 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 to loggerhead 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 (Kamrowski *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 (Kamrowski *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 (Kamrowski *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 hatchling success (Salmon *et al.*, 2000).

* Based on the reviewer comments and associated updates to the text, artificial lighting at the minor rookeries (10-100 nesters) of the nGBR is somewhat of a larger problem than initially considered by the SRT during its structured decision making efforts to determine extinction risk. We don't believe the revisions would appreciably change how the SRT evaluated risk for the DPS.

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 (un-quantified). 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 (M. Hamann, James Cook University, pers. comm., 2014).

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 (Marsh 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 off-shore 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 appears 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 analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels. The inadequacy of existing regulatory mechanisms for impacts to nesting 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 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).

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 (K. Dobbs, Queensland Parks Authority, pers. comm., as cited in Maison et al., 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 $\geq$ 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; Kwan and Bell, 2003; Chapman, 2003). 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 dienzofurans (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 that 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 turtle 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 Southwest 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). When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

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
- International Convention for the Prevention of Pollution from Ships (MARPOL)
- Ramsar Convention on Wetlands of International Importance
- Secretariat of the Pacific Regional Environment Program
- Torres Strait Treaty of 1978
- United Nations Convention on the Law of the Sea
- United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing
- United States Magnuson-Stevens Fishery Conservation and Management Act
- Western/Central Pacific Fisheries 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 GBR 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.

	Critical Assessment Elements					
	Element 1	Element 2	Element 3	Element 4	Element 5	Element 6
	Abundance (1 to 5)	Trends / Productivity (1 to 5)	Spatial Structure (1 to 5)	Diversity / Resilience (1 to 5)	Five-Factor Analyses (–2 to 0)	Conservation Efforts (0 to 2)
MEAN RANK	1.17	1.67	1.50	1.42	–0.67	0.58
SEM	0.17	0.19	0.26	0.19	0.19	0.23
RANGE	1–3	1–3	1–4	1–3	(-2)–0	0–2

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 quasi-extinction under current management regimes within 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.

	Probability Of Reaching Quasi-Extinction					
	<1%	1-5%	6-10%	11-20%	21-50%	>50%
MEAN ASSIGNED POINTS	72.50	9.08	10.58	5.42	2.42	0.00
SEM	10.74	2.27	5.20	3.39	1.43	0.00
Min	0	1	0	0	0	0
Max	99	21	60	40	15	0

Of the categories describing the probability that the Southwest Pacific DPS will reach quasi-extinction within 100 years, SRT member voted overwhelmingly in the '<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.

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 *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 Island, 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.

While the SRT determined the likelihood of reaching quasi-extinction 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.

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°N, 175°W to 9°N, 125°W to 40°S, 96°W to 40°S, 176°E, to 13°S, 171°E, and back to the 9°N, 175°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).

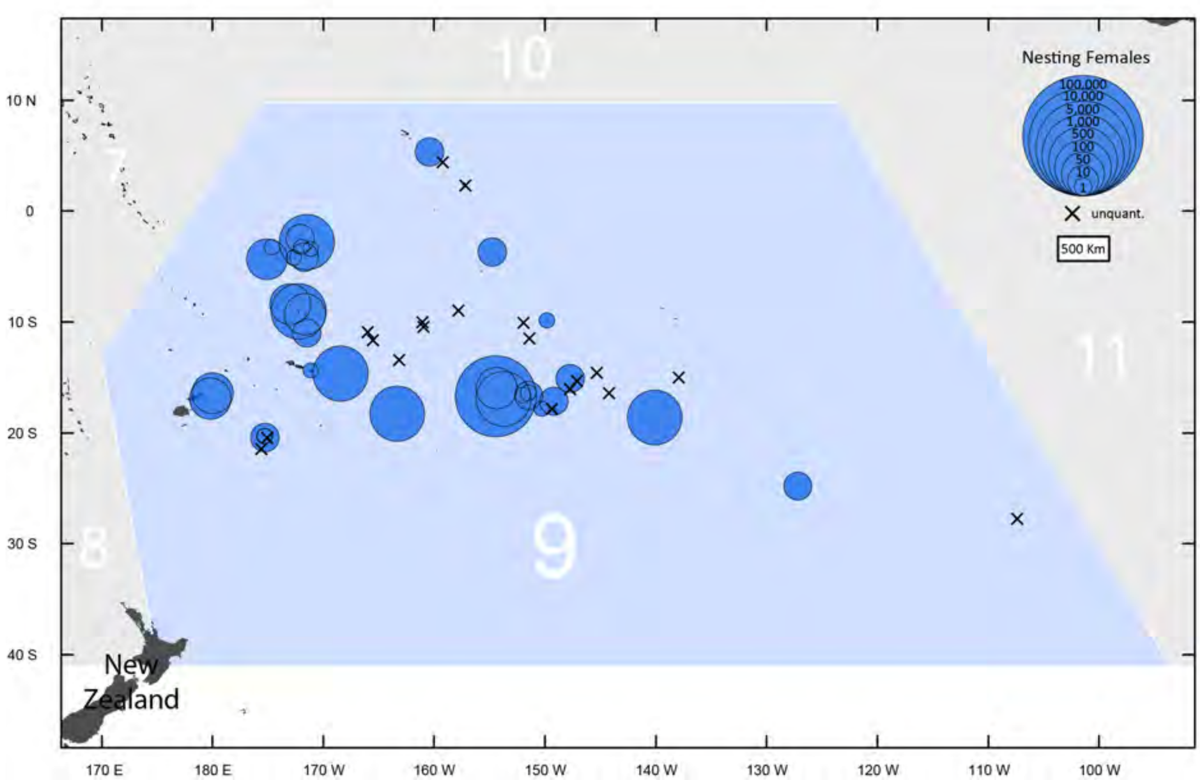

Figure 13.1. Nesting distribution of green turtles in the Central South Pacific DPS. Size of circles indicates estimated nester abundance (see Section 13.2.1). Locations marked with '×' indicate nesting sites lacking abundance information.

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). In 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 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 (Balazs, 1975a). 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 (Conservation International Pacific Islands Program, 2013). No sites have long-term monitoring programs, and no single site has had standardized surveys for even 5 continuous years. Most nesting areas are in remote, low-lying atolls that are logistically difficult to access. Unsurprisingly, many nesting areas (21 of 59, or 36 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 fewer than 700 nesters. When added to our 2,902 total, this DPS likely has fewer than 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. N/a indicates that recent nesting abundance not available.

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Am. Samoa (USA)	Rose Atoll	2006–2012	105
Am. Samoa (USA)	Swains Atoll	2007–2013	23
Am. Samoa (USA)	Tutuila	2007–2013	3
Cook Islands	Palmerston Atoll	2010	149
Cook Islands	Tongareva Atoll	2011-2012	172*
Cook Islands	Rakahanga	2009-2012	20*
Fiji	Nanuku Levu	2006	96
Fiji	Nukumbalati	2006	96
French Polynesia	Scilly Atoll	1991	1,050
French Polynesia	Mopelia	2010	168
French Polynesia	Motu One	1991	99
French Polynesia	Bora Bora	2010	33
French Polynesia	Tetiaroa	2010	25
French Polynesia	Tikehau	2007–2010	11
French Polynesia	Tupai	1995	6
French Polynesia	Maupiti	2010	3
French Polynesia	Maiao	2009	3
Kiribati	Enderbury	2002	129
Kiribati	Nikumaroro	2002	56
Kiribati	Canton	2002	29
Kiribati	Manra	2002	24
Kiribati	Tarawa	2007	17
Kiribati	Teraina (Washington)	1990s	15
Kiribati	Malden	1990s	15
Kiribati	Phoenix	2002	9
Kiribati	Orona	2002	8
Kiribati	Caroline	1990s	8
Kiribati	McKean	2000	6
Kiribati	Birnie	2002	5

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Tokelau	Nukunonu	1970s	210
Tokelau	Fakaofu	1970s	90
Tokelau	Atafu	1970s	60
Tonga	Fonuaika	2007	11
Tonga	Nukulei	2007	5
Tonga	Luanamo	2007	2
Tuvalu	Funafuti	2006	90
UK Overseas Territory	Henderson	1991	30

* This number was updated based on external review subsequent to SRT voting. We do not believe the revisions would appreciably change how the SRT evaluated risk for the DPS.

Table 13.2. The distribution of green turtle nester abundance in the Central South Pacific.

NESTER ABUNDANCE	# NESTING SITES DPS 9
Unquantified*	22
1–10	11
11–50	12
51–100	7
101–500	6
501–1000	0
1001–5000	1
5001–10000	0
10001–100000	0
>100,000	0
TOTAL SITES	**59**
TOTAL ABUNDANCE	**2,677**
% at LARGEST NESTING SITE	**36% (Scilly Atoll, French Polynesia)**

* Not included in Table 5.1

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, Tikehau, 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 the 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, 2012a), as far as 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, internesting 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, 2012a; 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 indicate 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.

All of these factors were considered given current management regimes. 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 USFWS, 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 (USFWS, 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 USFWS, 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, 2012b). 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, 2012a). 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 USFWS (1998) noted that degradation of coral reef habitats on the south side of Tutuila Island, American Samoa is occurring due 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 (USFWS, 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 (Obura and Stone, 2002).

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.

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, 2012a).

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

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.

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.

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 USFWS, 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 USFWS (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.* (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, 2002). 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'afeva, 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, 2012a). 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).

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 exultans*) were an issue at Rose Atoll prior to a 1993 eradication (USFWS, 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., USFWS, 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, 1975b). Balazs (1975b) 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, ghost crabs, Polynesian rats, 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, 2012a). Frigate birds 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. The analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels. 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. 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 green 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; GBR, 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 (2012a) 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, 1985a; 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; USFWS, 2012; Tagarino *et al.*, 2008). Marine debris is potentially hazardous to adults and hatchlings and is present in American Samoa at Rose Atoll (USFWS, 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 (NMFS and USFWS, 1998; Fish and Wildlife Service, 2012).

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 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, 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).

A recent study of 27 atoll islands in the central Pacific (including Kiribati and Tuvalu), for example, demonstrated that only 14 percent 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 percent) or increased in area (43 percent) 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. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

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 USFWS (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, 2012a). 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. comm., 2013; 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. comm.,). 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 Heemskereq 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).

The Phoenix Islands Protected Area (PIPA), with a size of 408,250 km^2 (157,626 sq. miles), 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. 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 Fakaofo 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 (USFWS), 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

Wallis and Futuna

No information could be found.

13.2.6.2. International Instruments

At least 16 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
- 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
- Indian Ocean Tuna Commission

- Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC)
- International Convention for the Prevention of Pollution from Ships (MARPOL)
- Inter-American Tropical Tuna Commission
- Memorandum of Understanding on Association of South East Asian Nations (ASEAN) Sea Turtle Conservation and Protection
- Secretariat of the Pacific Regional Environment Programme United Nations Convention on the Law of the Sea
- United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing
- United States Magnuson-Stevens Fishery Conservation and Management Act

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 and 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.

	Critical Assessment Elements					
	Element 1	Element 2	Element 3	Element 4	Element 5	Element 6
	Abundance (1 to 5)	Trends / Productivity (1 to 5)	Spatial Structure (1 to 5)	Diversity / Resilience (1 to 5)	Five-Factor Analyses (–2 to 0)	Conservation Efforts (0 to 2)
MEAN RANK	3.2	2.9	1.9	2.2	–1.3	0.5
SEM	0.3	0.3	0.2	0.3	0.2	0.2
RANGE	2–4	1–4	1–3	1–4	(–2)–0	0–2

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 quasi-extinction under current management regimes 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.

	Probability Of Reaching Quasi-Extinction					
	<1%	1-5%	6-10%	11-20%	21-50%	>50%
MEAN ASSIGNED POINTS	38.2	19.9	20.5	8.2	9.5	3.7
SEM	12.1	5.3	6.9	3.9	6.0	2.6
Min	0	0	0	0	0	0
Max	95	50	65	40	50	25

Of the six categories describing the probability that the Central South Pacific DPS will reach quasi-extinction 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.

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. 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˚E in the northwest corner, 41˚N, 143˚W in the northeast, 9˚N, 125˚W in southeast, and 9˚N, 175˚W in the southwest (Figure 14.1). 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, satellite-linked transmitter tracking, and genetic analyses (see Section 4).

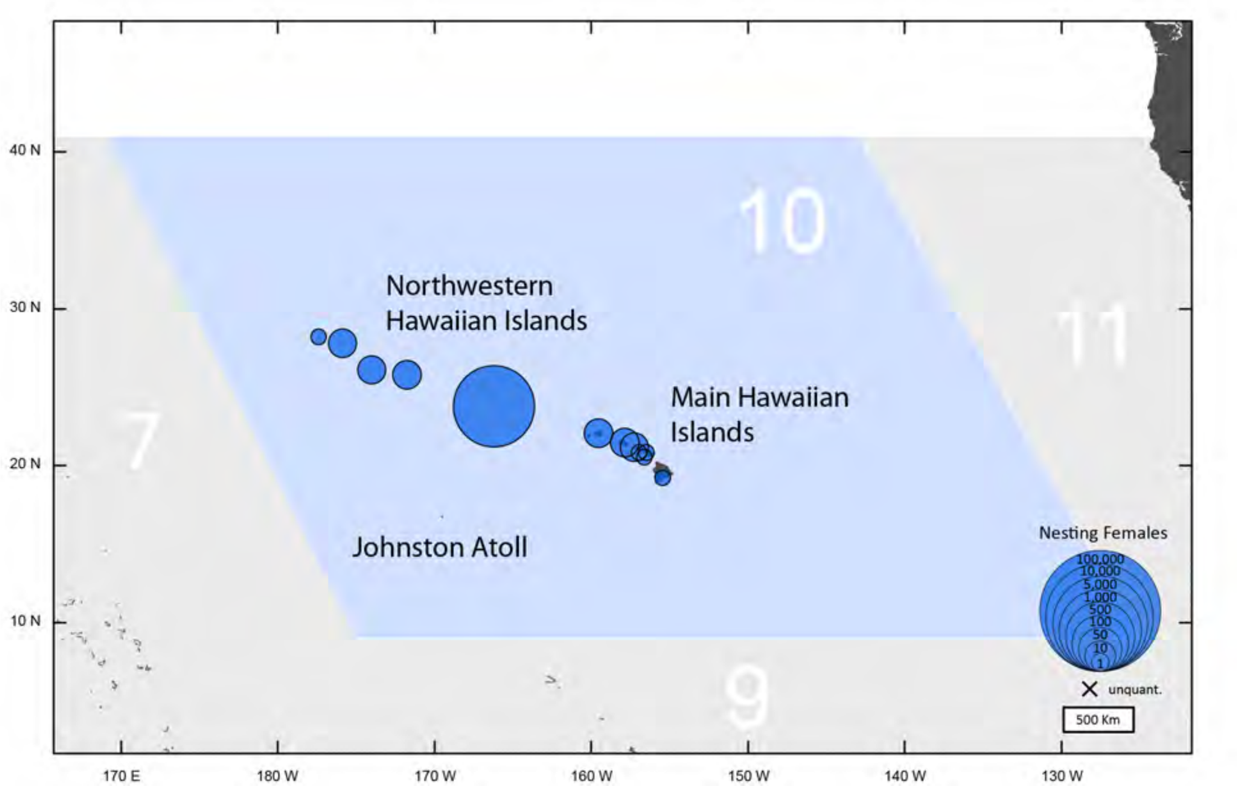

Figure 14.1. Geographic area of the Central North Pacific DPS. Size of circles indicates estimated nester abundance (see Section 14.2.1). DPS encompasses the entire Archipelago of Hawai'i and Johnston Atoll.

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 prominent focal point of green turtle nesting and hatchling 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 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.

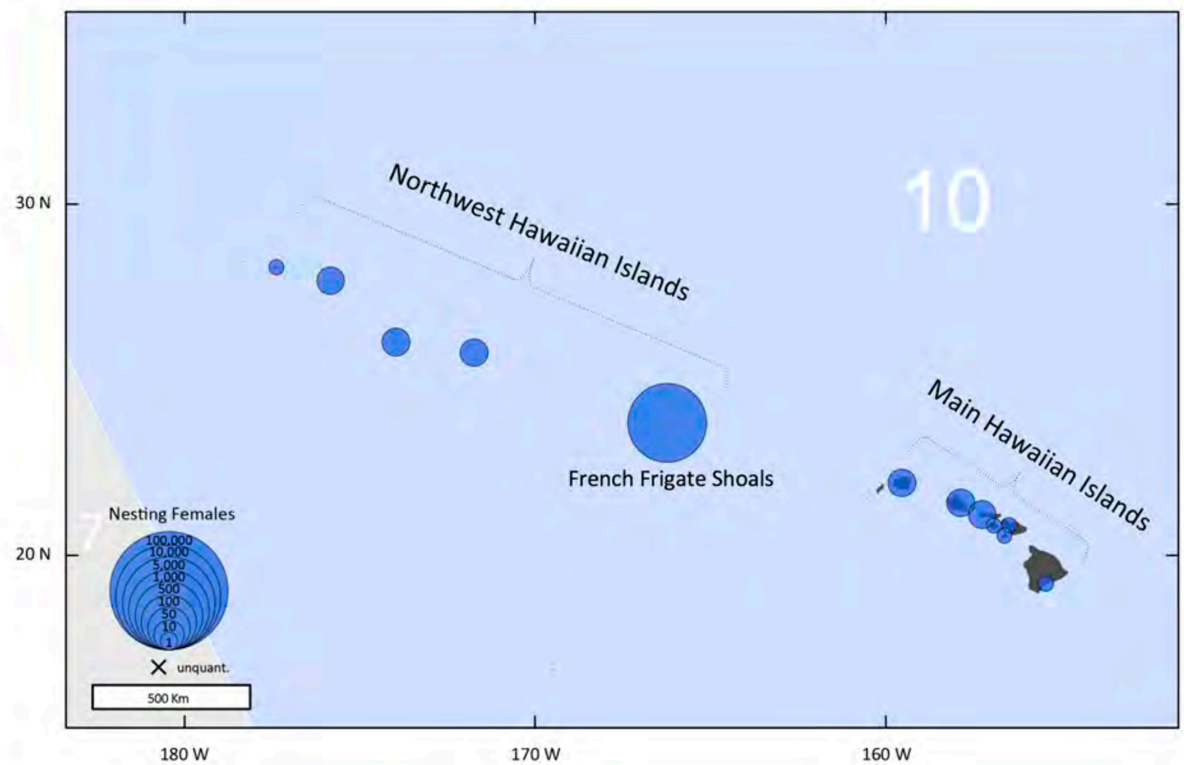

Figure 14.2. Closeup of nesting green turtles in the Central North Pacific DPS. Size of circles indicates estimated nester abundance (see Section 14.2.1).

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 Atoll. Since 2000, green turtle nesting on the MHI has emerged in low numbers on 7 islands (Frey *et al.*, 2013; Kittlinger *et al.*, 2013; PIFSC, unpublished data, 2013; Table 14.1).

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 14.1. Summary of green turtle nesting activity in the Central North Pacific DPS. Data are organized by country, nesting site, monitoring period, and estimated total nester abundance [(total counted females / year of monitoring) x remigration interval]. All nesting locations in this DPS have quantitative nesting estimates. For a list of references for these data, see Appendix 2.

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ESTIMATE	ESTIMATED NESTER ABUNDANCE
Hawai’i (USA)	Kure Atoll	n/a	unquantified
Hawai'i (USA)	Lana'i	2010-2012	1
Hawai'i (USA)	Kaho'olawe	2010-2012	1
Hawai'i (USA)	Hawai'i	2010-2012	1
Hawai'i (USA)	Midway Atoll	2011	3
Hawai'i (USA)	Maui	2010-2012	4
Hawai'i (USA)	O'ahu	2010-2012	11
Hawai'i (USA)	Lisianski Island	2011	15
Hawai'i (USA)	Kaua'i	2010-2012	16
Hawai'i (USA)	Laysan Island	2011	24
Hawai'i (USA)	Moloka'i	2011	24
Hawai'i (USA)	Pearl Hermes Reef	2011	36
Hawai'i (USA)	French Frigate Shoals	2009-2012	3,710

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 12 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 13 nesting sites, the smallest such number globally.

Table 14.2. The distribution of green turtle nester abundance in the Central North Pacific.

NESTER ABUNDANCE	# NESTING SITES DPS 10
unquantified	1
1-10	5
11-50	6
51-100	0
101-500	0
501-1000	0
1001-5000	1
5001-10000	0
10001-100000	0
>100,000	0
TOTAL SITES	**13**
TOTAL ABUNDANCE	**3,846**
PERCENT at LARGEST NESTING SITE	**96% (French Frigate Shoals, Hawai'i)**

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, 2004a, 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 Pala'au, 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).

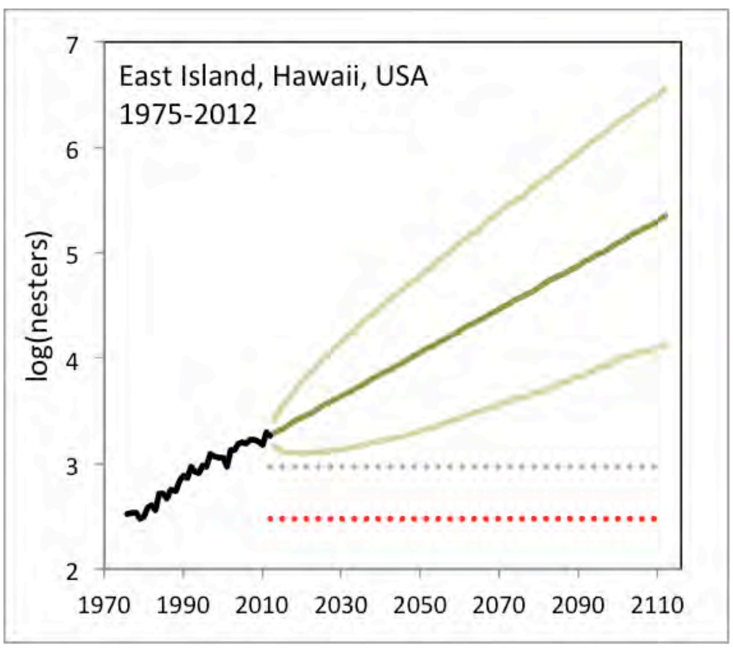

Figure 14.3. Stochastic Exponential Growth (SEG) Model Output for East Island, French Frigate Shoals, USA. Black line is observed data, green line is the average of 10,000 simulations, light green lines are the 2.5th and 97.5th percentiles (serving as the 95 percent credible interval), gray dotted line 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 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., Meffe *et al.*, 1994; Primack, 1998; Balazs and Kubis, 2007; Hunter and Gibbs, 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, 2004b; Wabnitz *et al.*, 2010). Nesters at the primary nesting area of FFS average 92.2 cm SCL, have an internesting 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 (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, 1975b). 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 diversity based on a total of four closely related mtDNA haplotypes identified (P. Dutton, NMFS, pers. comm., 2013).

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.

All of these factors were considered given current management regimes. 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, 1975b). 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, 1975b). Balazs and Chaloupka (2004a) 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 USFWS, 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; Friedlander *et al.*, 2006; Wedding and Friedlander, 2008; Wedding *et al.*, 2008; Van Houtan *et al.*, 2010). 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, Lana'I, Maui, and O'ahu have been degraded by sedimentation, sewage, or coastal construction (NMFS and USFWS, 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.*, 2005), 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 (http://www.fws.gov/refuges/profiles/index.cfm?id=12515).

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 the major 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 harvest (reported and estimated unreported) 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 previously in Hawai'i during a single day (Amerson *et al.*, 1974; Clapp and Wirtz, 1975; 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, 2014). 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, 2014). 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. Spirorchid (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). 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 depredation is unknown (Balazs, 1995). Sea birds may also prey on sea turtles in the marine environment (Balazs and Kubis, 2007). 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.

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.

14.2.5.4. Factor D: Inadequacy of Existing Regulatory Mechanisms

Regulatory mechanisms are in place that are designed to address direct and incidental take of green turtles in the Central North Pacific DPS. The analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels. Impacts to the nesting beach habitat and marine habitat (Factor A), overutilization (Factor B), predation (Factor C), and fishery bycatch (Factor E) continue throughout the DPS to varying degrees.

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 comprehensiveprotection and recovery activities that have 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 Hawai'i, they are also protected by the Hawai'i Revised Statutes, Chapter 195D (Hawai'i State Legislature, accessed 9/10/2010) and Hawai'i Administrative Rules, 13-124 (Hawai'i 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 Hawai'i. If this DPS did not remain listed under the ESA, it is unclear whether or not it would remain listed under the Hawaiian statute and/or what state protections and management would remain in place. 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).

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, international treaties often do not realize their full potential, either because they do not include all key countries, do not specifically address sea turtle conservation, are handicapped by the lack of a sovereign authority that promotes enforcement, 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), and constitute a much greater fishing effort (NMFS, 2012). While exact numbers are not available, it is estimated that, at a minimum, 100 green turtles from the Central North Pacific DPS are captured and killed annually by longline bycatch (NMFS, 2012).

Gill Net Fisheries

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 does occur (NMFS, 2012). Each year green sea turtles are incidentally entangled in net gear, some of these resulting in mortality (e.g., Francke, 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.* (2008) report 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, 1985a; 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, 2013, 2014). For example, thirty-six green sea turtles stranded in Hawaii in 2012 (Francke, 2013) and forty-two in 2013 (Francke, 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.

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, and petroleum contamination of turtle foraging habitat (Balazs, 1985b). 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 Hawai'i. At least 11 green turtles were recorded as having been struck by boats in 2012 (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). Thirteen reported vessel groundings have occurred in the NWHI in the last 60 years (Keller *et al.,* 2009). While we do not have a number for vessel groundings in the MHI, vessel activity occurring in or around green sea turtle foraging habitat in the MHI islands is much greater than in the NWHI. Vessel traffic and presence can also have negative effect through habitat damage from anchors, waste discharge, light and noise (Keller *et al.*, 2009).

14.2.5.5.4. 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, and that one formerly significant nesting site – Whale-Skate Island – is now completely submerged. 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 hatchling 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 hatchling sex determination and sex ratios (Balazs and Kubis, 2007). Sand temperatures prevailing during the middle third of the incubation period determine the sex of hatchling 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 hatchling 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. Possible changes to ocean currents and dynamics may 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 if nesting in the MHI increases, the importance of the threat from these potential predators would increase.

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 hatchling 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. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

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. Both on-the-ground conservation actions and financial and other resources have resulted in significant population growth of green turtles.

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 USFWS'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

At least 16 international treaties and/or 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
- Inter-American Tropical Tuna Commission
- 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 Convention on the Law of the Sea
- 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 parts 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 fraction 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 where 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.

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 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.

	Critical Assessment Elements					
	Element 1	**Element 2**	**Element 3**	**Element 4**	**Element 5**	**Element 6**
	Abundance (1 to 5)	**Trends / Productivity (1 to 5)**	**Spatial Structure (1 to 5)**	**Diversity / Resilience (1 to 5)**	**Five-Factor Analyses (–2 to 0)**	**Conservation Efforts (0 to 2)**
MEAN RANK	2.7	1.2	2.9	2.7	-0.9	0.6
SEM	0.3	0.1	0.4	0.3	0.2	0.2
RANGE	1–4	1–2	1–4	1–5	(-2)–0	0–2

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 quasi-extinction under current management regimes 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.

	Probability Of Reaching Quasi-Extinction					
	<1%	**1-5%**	**6-10%**	**11-20%**	**21-50%**	**>50%**
MEAN ASSIGNED POINTS	49.1	16.3	14.9	7.5	7.3	4.9
SEM	11.6	3.8	4.5	3.3	4.4	3.5
Min	0	2	0	0	0	0
Max	98	40	50	35	52	39

Of the six categories describing the probability that the Central North Pacific DPS will reach quasi-extinction 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. 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.

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.

	Critical Assessment Elements					
	Element 1	**Element 2**	**Element 3**	**Element 4**	**Element 5**	**Element 6**
	Abundance (1 to 5)	**Trends / Productivity (1 to 5)**	**Spatial Structure (1 to 5)**	**Diversity / Resilience (1 to 5)**	**Five-Factor Analyses (–2 to 0)**	**Conservation Efforts (0 to 2)**
MEAN RANK	2.67	1.33	3.25	2.67	-0.92	0.58
SEM	0.28	0.19	0.37	0.45	0.23	0.19
RANGE	1–4	1–3	1–5	1–5	(-2)–0	0–2

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 96 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 quasi-extinction under current management regimes, 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.

	Probability of Reaching Quasi-Extinction					
	<1%	**1-5%**	**6-10%**	**11-20%**	**21-50%**	**>50%**
MEAN ASSIGNED POINTS	46.25	17.17	15.50	7.92	7.83	5.33
SEM	11.76	4.06	4.62	3.38	4.16	3.89
Min	0	1	0	0	0	0
Max	98	40	50	35	49	44

With respect to the overall risk of extinction, or the six categories describing the probability that the Central North Pacific DPS will reach quasi-extinction 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%' risk of extinction, and a 54 percent vs. 51 percent chance that the population has '>1%' 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.

Votes on both the importance placed on critical elements and extinction risk varied widely among SRT members. Overall, 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, 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 border, USA (42˚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 143˚W and 96˚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).

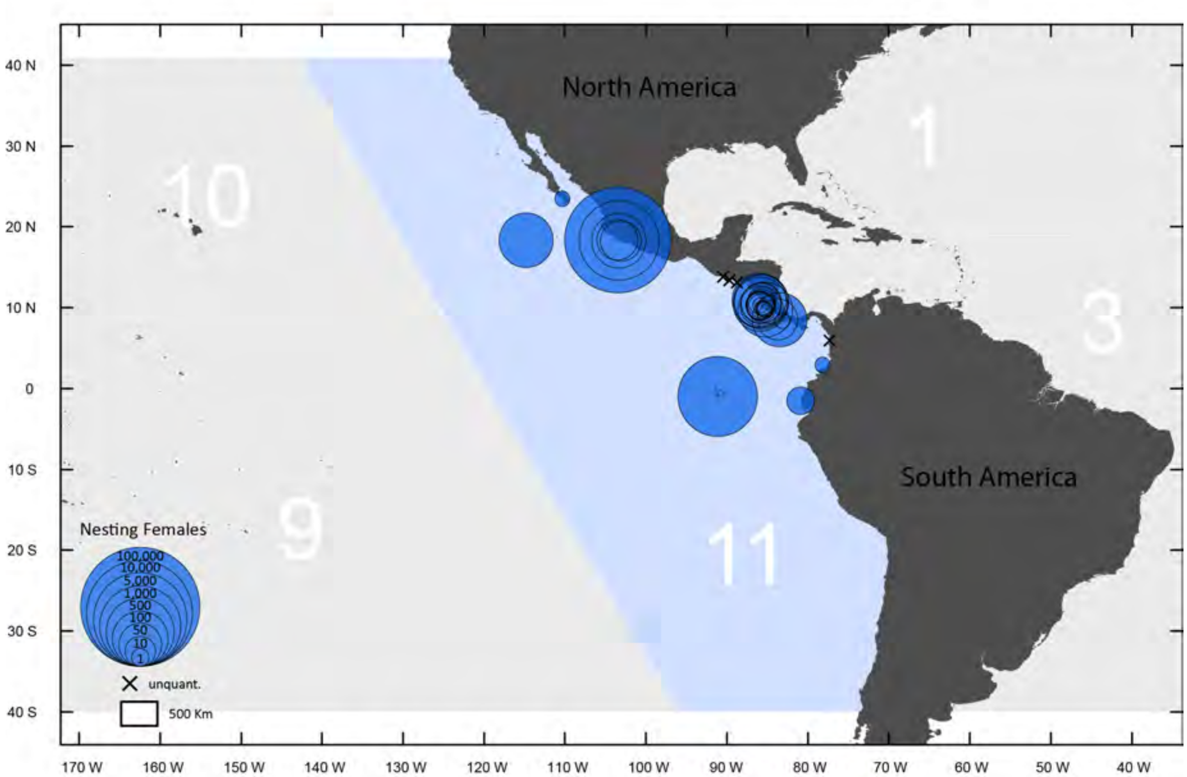

Figure 15.1. Nesting distribution of green turtles in the East Pacific DPS (blue-shaded area marked with '11'). Size of circles indicates estimated nester abundance (see Section 15.2.1). Locations marked with '×' indicate nesting sites lacking abundance information.

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 39 total nesting sites for which abundance information is available. There are sporadic nesting events in many other areas in the East Pacific DPS, such as Guatamala and Peru, 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, bears 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, Motin 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 (Juarez-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 (Green and Ortiz-Crespo, 1995; M. Hurtado, Hurtado and Associates, Inc., unpubl. data). From 2001 to 2002, 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. Playa San José-Bat Islands and Playa Colorada together host a minimum of 650 females, based on short-term observations (L. Fonseca, WIDECAST, unpubl. data; M. Heidermeyer, U. Costa Rica, Unpubl. Data). Nombre de Jesús-Zapotillal Beaches had 1,199 females nest there from 2012 to 2014 (E. Vélez-Carballo, Asociación Kuemar, and R. Piedra-Chacón, Minae-SINAC, unpubl. data), and Cabuyal Beach hosted 273 nesters in 2012 alone (P. Santidrián-Tomillo, The Leatherback Trust, pers. comm., 2012) Numerous other beaches host ≤10–50 nests/yr (Table 15.1).

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. Estimated total nester abundance for the sub-population is calculated as: [(Total Counted Females/Years of Monitoring) x Remigration Interval] and is calculate only for those sites for which there is sufficient data to estimate abundance: valued denoted with a "*". Remigration interval for green turtles in the East Pacific is calculated at 3 yrs (Alvarado-Díaz and Figueroa, 1990). For sites with short-term (≤1 yr) data available, values represent nester counts for the time period only, and thus represent the miminum total nester abundance for the site. For a list of references for these data, see Appendix 2.

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ABUNDANCE	ESTIMATED NESTER ABUNDANCE
Mexico	Colola, Michoacán	2010–12	11,588*
Mexico	Llorona, Michoacán	2007	90
Mexico	Bahia Maruata, Michoacán	2007	1,149
Mexico	Motin de Oro, Michoacán	2007	240
Mexico	Arenas Blancas, Michoacán	2007	90
Mexico	Cape Region, BCS	2007–09	7
Mexico	Revillagigedos, MX	November-December 2008	500
Costa Rica	Playa Junquillal	June 2012-March 2013	48
Costa Rica	Playa San José, Bat Islands	November-March 2013	450
Costa Rica	Playa Coloradas	4 observations, 2 in January 2013, 2 in March 2013	200
Costa Rica	Playa Nancite	2012	20
Costa Rica	Playa Naranjo	2010-2012, 2014	80
Costa Rica	Playa Cabuyal	2012	273
Costa Rica	Playa Nombe de Jesus – Zapotillal	2012-2014	1,199*
Costa Rica	Playa Prieta, Playa Virador, Playa Matapalo, Playa Blanca (4 sites combined)	1 time observation January 2010	70
Costa Rica	Playa Grande -Ventanas	Year round	48
Costa Rica	Playa Langosta	November-March 2013	48

COUNTRY	NESTING SITE	MONITORING PERIOD USED FOR NESTER ABUNDANCE	ESTIMATED NESTER ABUNDANCE
Costa Rica	Playa Avellanas	October-March 2012	9
Costa Rica	Playa Lagartillo	October-March 2012	48
Costa Rica	Playa Callejones	October-March 2012	48
Costa Rica	Playa Blanca	October-March 2012	48
Costa Rica	Playa Junquillal	Year round	48
Costa Rica	Playa Ostional	October-April 2012	48
Costa Rica	Playa Buena Vista	July-December 2009	9
Costa Rica	Playa Camaronal	2012	48
Costa Rica	Playa Corozalito	June-December 2012	9
Costa Rica	Playa San Miguel	2012	9
Costa Rica	Playa Caletas	June-December 2012	9
Costa Rica	Punta Banco	June-December 2010	9
Colombia	Isla Gorgona	2007–2009	4
Ecuador	Galapagos (4 beaches)	2003–2005	3,603*
Ecuador	Mainland	2010	15

Table 15.2. Green turtle nester abundance distribution in the East Pacific DPS.

NESTER ABUNDANCE	# NESTING SITES DPS 11
Unquantified*	4
1 to 10	8
11–50	10
51–100	7
101–500	6
501–1000	0
1001–5000	3
5001–10000	0
>10000	1
TOTAL SITES	**39**
TOTAL NESTER ABUNDANCE	**20,062**
PERCENTAGE at LARGEST NESTING SITE	**58% (Colola, Mexico)**

* Not included in Table 15.1

15.2.2. Population Trends

There are two sites in the Eastern Pacific, Galapagos, Ecuador and Colola, Mexico, for which some level of knowledge is available regarding nesting trends. Only one of these—Colola—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.

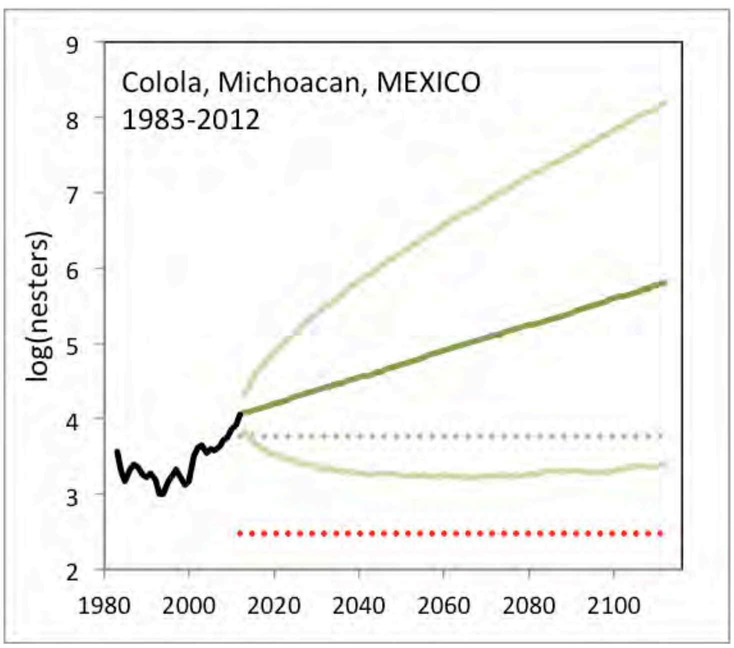

Figure 15.2. Stochastic Exponential Growth (SEG) Model Output for Colola, Michoacan, Mexico. Black line is observed data, dark green line is the average of 10000 simulations, green lines are the 2.5th and 97.5th percentiles, grey dotted line is trend reference, and red dotted line is absolute abundance 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 (Cliffton *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 East 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. Forty-nine 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 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, internesting 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 Galapagos is 46.0 (Zárate *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 (Zárate *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 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 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.

All of these factors were considered given current management regimes. 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, the nutrient flow and structure within sea grass communities in many coastal areas are likely modified today due to the depletion of green turtles, which during times of higher abundance, would have been keystone consumers in these habitats (Bjorndal, 1980; Thayer et al., 1992; Seminoff et al., 2012b). 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.*, 2012a). 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 USFWS, 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. The analysis of these existing regulatory mechanisms assumed that all would remain in place at their current levels. 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, 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" around natural floating objects or Fish Aggregation Devices –FADs- (Hall 1998) because turtles associate with these floating objects presumably looking for shelter. Pelagic longlines in the eastern Pacific Ocean are used to capture species like tunas, swordfish, billfishes, *mahi-mahi* and sharks. All species of sea turtles that occur in the region interact with longline gear in the EP, but species frequency, bycatch and mortality rates vary spatially and seasonally (Kelez *et al.,* 2008).

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, 2007).

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. When assessing conservation efforts, we assumed that all conservation efforts would remain in place at their current levels.

The first of these conservation alliances commenced in 1997 when, after years of information exchange about shared populations among 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 Presidental 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 (Aridjis, 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 in the United States that promote the protection and conservation of sea turtles. The most relevant to sea turtle protection is 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. Both on-the-ground conservation actions, as well as financial and other resources, have resulted in significant population growth of green turtles.

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

At least 10 international 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).

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.

	Critical Assessment Elements					
	Element 1	**Element 2**	**Element 3**	**Element 4**	**Element 5**	**Element 6**
	Abundance (1 to 5)	**Trends / Productivity (1 to 5)**	**Spatial Structure (1 to 5)**	**Diversity / Resilience (1 to 5)**	**Five-Factor Analyses (–2 to 0)**	**Conservation Efforts (0 to 2)**
MEAN RANK	1.60	1.20	1.70	1.70	–0.80	1.00
SEM	0.27	0.13	0.26	0.30	0.13	0.21
RANGE	1–3	1–2	1–3	1–4	(–1)–0	0–2

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 is was unlikely that these element contributes significantly to risk of extinction by their selves, 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 15.4. Summary of Green Turtle SRT member expert opinion about the probability that the DPS will reach quasi-extinction under current management regimes 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.

	Probability of Reaching Quasi-Extinction					
	<1%	1–5%	6–10%	11–20%	21–50%	>50%
MEAN ASSIGNED POINTS	63.64	16.73	6.36	7.82	5.00	0.45
SEM	11.97	6.97	2.66	5.27	3.50	0.45
Min	0	0	0	0	0	0
Max	97	65	30	50	35	5

Of the six categories describing the probability that the East Pacific DPS will reach quasi-extinction within 100 years (Table 15.4; Figure 15.3), 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.

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 depredation 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 quasi-extinction 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 or less. It is important to note, however, our analysis did not consider the scenario in which current laws or regulatory mechanisms were not continued. Given the conservation dependence of the species, without mechanisms in place to continue conservation efforts and funding streams in this DPS, some threats could increase and population trends could be affected.

16. SYNTHESIS

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 16.1. Summary of the spatial separation, demography, tagging and genetics used to determine discreteness.

DISCRETENESS				
DPS	**Spatial Separation (*physical*)**	**Demography *(physiological)***	**Tagging *(behavioral)***	**Genetics**
1. North Atlantic	Some overlap at southern edge of N Atl range w/ DPS 3; no overlap with DPS 2		Minimal transboundary recoveries (some w/ DPS 3, no transboundary tag recoveries w/ DPS 2); localized movements; distinct FP phylogeny compared to DPS 3	N Atl haplotypes found juveniles captured in Brazil and Argentina (DPS 3); no genetic structure from nDNA w/ DPS 3, but a small number of genetic markers were examined
2. Mediterranean	Only population in entire sea basin	Second smallest MNS of any region (after EP)	No transboundary recoveries w/ DPS 1; localized movements; no immigration from DPS 1 despite extensive data	Clear genetic differences w/ DPS 1
3. South Atlantic	Some overlap at northern edge of range w/ DPS 1	Largest MNS globally	Extensive movements within region, but no immigration or emigration revealed through satellite telemetry w/ DPS 1 or 4; distinct FP phylogeny compared to DPS 1	Haplotype frequencies provide no evidence for contemporary connectivity around cold waters of Cape of Good Hope; haplotypes from turtles in SW S Africa (Mozambique Channel) are from the same clade as those in S. Atlantic, but this reflects distant evolutionary history, high local connectivity

DISCRETENESS				
DPS	**Spatial Separation (*physical*)**	**Demography *(physiological)***	**Tagging *(behavioral)***	**Genetics**
4. SW Indian	Cape of Good Hope separates from DPS 3; no clear current boundaries w/ DPS 5 or 6, Apparent nesting gap w/ DPS 3, 5, 6	MNS larger than DPS 5 or 6	Minimal transboundary recoveries w/ DPS 5; no transboundary recoveries w/ DPS 3; no transboundary recoveries and minimal data w/ DPS 6, localized movements despite extensive data; no immigrations but minimal data (DPS 6)	Genetic differences present but not strong (DPS 6), no nDNA, just mtDNA (DPS 6), Strong genetic differences (DPS 3,5)
5. N Indian	Apparent nesting gap w/ DPS 4, 6	MNS smaller than DPS 4, 6	Minimal transboundary recoveries w/ DPS 4; no transboundary recoveries w/ DPS 6; localized movements	Globally unique clade; clear genetic differences w/ DPS 4, 6; almost all rookeries in N Indian un-sampled
6. E Indian- W Pacific	Wallace Line is biogeographic boundary w/ DPS 7); apparent nesting gap w/ DPS 5; large distance from DPS 4; oceanographic currents suggest possible connectivity w/ DPS 4	MNS larger than DPS 5	Moderate transboundary recoveries w/ DPS 7, 8; rare transboundary recoveries w/ DPS 9; localized movements; no transboundary recoveries w/ DPS 10	Globally unique haplotypes; distinct and high nucleotide diversity; clear genetic differences from DPS 5, 7-9; historical genetic connectivity w/ DPS 4

DISCRETENESS				
DPS	**Spatial Separation (*physical*)**	**Demography *(physiological)***	**Tagging *(behavioral)***	**Genetics**
7. CW Pacific	Wallace Line - biogeographic boundary w/ DPS 6; oceanographic boundary w/ DPS 10		Moderate transboundary movements, although small sample size; moderate transboundary recoveries w/ DPS 6; minimal transboundary recoveries w/ DPS 8	Globally unique haplotypes, clear genetic differences w/ DPS 6, 8, 9; AMOVA supports stand-alone entity; no genetic immigration from DPS 8 or 9
8. SW Pacific	Closely proximate DPSs		Moderate transboundary recoveries w/ DPS 6; minimal transboundary recoveries w/ DPS 7, 9; localized movement, although small sample size	Globally unique haplotypes; oldest haplotype lineages; distinct and high nucleotide diversity; clear genetic differences w/ DPS 6, 7, 9; nDNA and mtDNA distinctiveness w/ DPS 6, 7, 9
9. CS Pacific	Oceanographic barrier w/ DPS 10, 11; EP turtles found in Am Samoa longline (DPS 11)	Data deficient	Localized movements although limited data; modest transboundary recoveries w/ DPS 7, 8; minimal tranboundary immigration from DPS 7, 8; no recoveries w/ DPS 10	Clear genetic differences w/ DPS 7, 8, 10, but only two rookeries sampled
10. CN Pacific	Most isolated archipelago globally; oceanographic barrier w/ DPS 11; large distances to DPS 7-9, 11	MNS larger than DPS 11	Rare transboundary recoveries but extensive data w/ DPS 6-9, 11; localized movements w/ extensive data	No shared haplotypes w/ DPS 6-9; shared haplotype with DPS 11

DISCRETENESS				
DPS	**Spatial Separation (*physical*)**	**Demography *(physiological)***	**Tagging *(behavioral)***	**Genetics**
11. Eastern Pacific	Moderate numbers of juveniles found in DPS 7, 8 and high seas of CNP (DPS 10); CSP (DPS 9) 'yellow' juveniles found in southeastern EP	Smallest MNS of any DPS; mostly black in color	No tag recoveries or satellite tracks of EP turtles outside EP, although small number of EP turtles found in DPS 7, 8 and10; no tag recoveries or satellite tracks of turtles from other DPSs in EP, although small number of turtles from DPS 9 found in EP	Clear genetic differences w/ DPS 7-10; some shared haplotypes w/ DPS 10

Table 16.2. Summary of the ecological setting, gap in range, and marked genetics used to determine significance.

SIGNIFICANCE				
DPS	**Ecological Setting**	**Gap in Range**	**Marked Genetics**	**Other**
1. North Atlantic	Caribbean sea unique w/ expansive seagrass beds, broad continental shelf; Nesting from N. FL to NC outside normal latitudinal range	No gene flow w/ Med (DPS 2); some gene flow with DPS 3	Distinct genetic differences based on mtDNA (DPS 2,3); some globally unique haplotypes	2-year remigration interval; high incidence of FP
2. Mediterranean	Unique habitat as enclosed sea, oligotrophic, low-productivity waters, most saline waters and northern-most nesting	Population encompasses large region; apparent biogeographic boundary of W. Med would hinder re-population	100% globally unique haplotypes; significant difference in mtDNA markers from DPS 1	Second smallest MNS of any region (after EP); northern-most latitude for nesting
3. South Atlantic	Ascension Isl. is only mid-ocean ridge island nesting site	Population encompasses vast region (southern hemisphere of ocean basin)	Globally unique haplotypes	Largest MNS globally
4. SW Indian	Major cold water upwelling in the Mozambique Channel creates distinctive habitat	No known immigration from DPS 3, 5, 6; apparent biogeographic barrier w/ DPS 3		Largest MNS for Indian Ocean
5. N Indian	Unique habitat w/ heat adapted coral in Persian and Red Seas; highly saline waters	Isolated and far from adjacent DPSs (4 and 6)	Limited genetic data from one nesting population shows globally unique and very divergent haplotypes in Saudi Arabia	

SIGNIFICANCE				
DPS	**Ecological Setting**	**Gap in Range**	**Marked Genetics**	**Other**
6. E Indian- W Pacific	Most extensive continental shelf globally; high rainfall and extensive river runoff produce low salinity water in the N Indian Ocean	Population encompasses large region; loss would create major connectivity gap between DPSs 4-5 and 7-8	Ancestral haplotypes; significant mtDNA diversity	
7. CW Pacific		Apparent oceanic boundary w/ DPS 10; apparent biogeographic boundary w/ DPS 6	Globally unique haplotypes	
8. SW Pacific	GBR provides unique habitat; periodic isolation over geological time		Ancient lineage; significant mtDNA diversity	
9. CS Pacific	Nesting on small atolls and islands and more spread out than elsewhere (no nesting stronghold)	Population encompasses large oceanic region; apparent oceanic boundary w/ DPS 10	A single, globally unique haplotype; extensive sampling in other regions has not detected haplotype	
10. CN Pacific	No continental shelf, only mid-basin oceanic pinnacles	Encompasses large oceanic region; most isolated of all DPSs; apparent biogeographic boundary w/ DPS 11 and oceanic boundary w/ DPS 7, 9	Globally unique haplotypes; extensive sampling in other regions has not detected haplotypes; historic gene flow w/ DPS 11	High incidence of FP; basking

SIGNIFICANCE				
DPS	**Ecological Setting**	**Gap in Range**	**Marked Genetics**	**Other**
11. Eastern Pacific	Unique diet due to very narrow continental shelf and low levels of seagrass; equatorial upwelling (ENSO)	Very large range; apparent biogeographic boundary w/ DPS 10	Globally unique haplotypes; extensive sampling in other regions has not detected haplotypes; historic gene flow w/ DPS 10	Smallest MNS of all regions; unique overwintering behavior; basking in Galapagos

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.

	NUMBER OF NESTING SITES											GLOBAL
NESTER ABUNDANCE	**N Atlantic**	**Mediterranean**	**S Atlantic**	**SW Indian**	**N Indian**	**E Indian / W Pacific**	**CW Pacific**	**SW Pacific**	**CS Pacific**	**CN Pacific**	**E Pacific**	**TOTALS**
unquantified	26	0	37	23	1	7	16	1	22	1	4	**138**
1-10	17	21	0	1	5	7	6	0	11	5	8	**80**
11-50	6	5	0	0	5	8	9	0	12	6	10	**62**
51-100	3	3	2	0	0	3	6	0	7	0	7	**32**
101-500	10	3	3	3	15	11	12	2	6	0	6	**71**
501-1000	4	0	3	1	4	8	0	3	0	0	0	**22**
1001-5000	6	0	3	4	5	7	2	3	1	1	3	**36**
5001-10000	0	0	1	2	1	5	0	0	0	0	0	**8**
10001-100000	1	0	2	4	2	1	0	3	0	0	1	**14**
>100000	1	0	0	0	0	0	0	0	0	0	0	**1**
TOTAL # SITES in DPS	74	32	51	37	38	57	51	12	59	13	39	**464**
TOTAL NESTER ABUNDANCE	167,528	404-992	63,332	91,159	55,243	77,009	6,518	83,058	2,677	3,846	20,062	**563,826-564,464**
LARGEST NESTING SITE	Tortuguero, Costa Rica	Akyatan, Turkey	Poilão, Guinea Bissau	Europa Island	Ras Sharma, Yemen	Wellesey Group, Australia	FSM	nGBR, Australia	Scilly Atoll, Fr. Polynesia	French Frigate Shoals, Hawai'i	Colola, Mexico	
% at LARGEST NESTING SITE	79%	25%	46%	30%	33%	32%	22%	37%	36%	96%	58%	

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.

DPS	Nesting Site	Bar plot / PVA	Population Trend
1. North Atlantic	El Cuyo, Mexico	Bar plot	▬
	San Felipe, Cuba	Bar plot	▬
	Guanal, Cuba	Bar plot	▬
	Tortuguero, Costa Rica	PVA	▲
	Isla Aquada, Mexico	PVA	▲
	Guanahacabibes, Cuba	PVA	▲
	Index Beach, Florida	PVA	▲
2. Mediterranean	Akrotiri, Cyprus	Bar plot	?
	North Karpaz, Cyprus	Bar plot	▬
	Akyatan, Turkey	Bar plot	▬
	Kazanli, Turkey	Bar plot	▼
	Israel	Bar plot	▲
	Samandag, Turkey	Bar plot	?
	West Coast, Cyprus	PVA	▼
3. South Atlantic	Ascension Island, UK	Bar plot	?
	Galibi Reserve and Matapica, Suriname	Bar plot	▬
	Atol das Rocas, Brazil	Bar plot	▬
4. Southwest Indian	Glorieuses, Eparses Islands, France	Bar plot	▬
	Europa, Eparses Island, France	Bar plot	▬
	Tromelin, Eparses Islands, France	Bar plot	▬

DPS	Nesting Site	Bar plot / PVA	Population Trend
5. North Indian	Daran Beach, Jiwani, Pakistan	Bar plot	▼
	Zabargard, Egypt	Bar plot	
6. East indian-West Pacific	Wan-an, Taiwan	Bar plot	▼
	Lanyu, Taiwan	Bar plot	—
	Sabah Turtle Islands, Malaysia	PVA	▲
	Royal Navy Center, Thailand	PVA	▼
	Redang, Terrengganu, Malaysia	PVA	▼
	Thameehla Island, Myanmar	PVA	▼
7. Central West Pacific	Chichijima, Japan	PVA	▲
8. Southwest Pacific	Raine Island, Australia	PVA	▲
	Heron Island, Australia	PVA	▲
9. Central South Pacific	not available		?
10. Central North Pacific	East Island, Hawaii	PVA	▲
11. East Pacific	Colola, Mochoacan, Mexico	PVA	▲

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.

	Spatial Structure		
DPS	**Genetic data**	**Flipper/satellite tagging**	**Demographic data**
1. North Atlantic	Shallow population structuring		Low population structuring
2. Mediterranean	No population structuring	Similar migration pattern	Consistent parameters, small nesting turtles
3. South Atlantic	Shared haplotype, strong reproductive isolation from other nesting sites, shared haplotype with foraging N. Atlantic	Transoceanic developmental migrations, the wide range of the DPS and the interconnectedness of the different regions	Vary widely among nesting sites, substantial spatial structuring
4. Southwest Indian	Moderate spatial structuring	Green turtles nesting along the East African coast confine their migration to along the coast.	
5. North Indian	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	Foraging within Indian ocean	Varies, substantial spatial structuring
6. East Indian-West Pacific	Complex population structure, few common and widespread haplotypes	Broad migration distribution and numerous potential foraging areas	Vary widely among nesting sites, substantial spatial structuring
7. Central West Pacific	Nesting sites separated by more than 1,000 km were significantly differentiated from each other	Nesting females migrate to areas within and outside of the Central West Pacific	Variation suggests substantial spatial structuring
8. Southwest Pacific	Substantial spatial structuring	Foraging is widely dispersed throughout this DPS and also into other DPSs	
9. Central South Pacific	Substantial spatial structuring	Post-nesting females travel the complete geographic breadth of this DPS	No structuring of traits within the DPS

	Spatial Structure		
10. Central North Pacific	Low level of spatial structuring	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	No structuring of traits within the DPS
11. East Pacific	Substantial spatial structuring	Track clustering in NW Mexico to Southern United States, and in the SE Pacific, from the Galapagos Islands to the high seas	Regional variation

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.

	Diversity / Resilience			
	Spatial range	**Nesting season**	**Nest Site**	**Genetic Diversity**
1. North Atlantic	Widespread	Similar	Continental and island	Shallow regional substructuring
2. Mediterranean	Limited	Similar	Insular and continental	Low population substructuring
3. South Atlantic	Widespread	Varies	Continental and island	Shallow structuring and are all dominated by a common shared haplotype
4. Southwest Indian	Limited	Year-round with peaks that vary	Mostly islands, atolls, and mainland of Africa	High diversity and a mix of unique and rare haplotypes, as well as common and widespread haplotypes
5. North Indian	Moderately dispersed	Varies	Continental and island	Limited sampling of single rookery very distinct from other rookeries elsewhere in the Indian Ocean
6. East Indian-West Pacific	Widespread	Varies	Continental and island	Varying levels of spatial structure characterized by the presence of rare/unique haplotypes at most rookeries
7. Central West Pacific	Moderately dispersed	Varies	Various islands and atolls	
8. Southwest Pacific	Widely dispersed throughout the region	similar- year-round with peak	Coral and rocky reefs, sea grass meadows and algal turfs on sand and mud flats	High genetic diversity
9. Central South Pacific	Widely dispersed	Varies	Low-lying coral atolls or oceanic islands	Moderate level of diversity and presence of unique haplotypes
10. Central North Pacific	Limited	similar	Low-lying coral atoll	Low level of stock substructuring
11. East Pacific	Limited	Differ within DPS	Substantial nesting at both insular and continental nesting sites, high sloped, shaded	Presence of rare/unique haplotypes

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.

North Atlantic				
	KNOWN THREATS	EXTENT	LIFE STAGE AFFECTED	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal Development with lighting (including armoring, jetties)	2	1, 2, 3	3, 6
	Erosion from storm events and sand mining	1	1, 2	3
	Beach engineering	2	1, 2	2, 6
	Climate change: Sea level rise and increased storm events- loss of habitat	1	1, 2	3
	Beach Driving	3	1, 2	2
	Fishing practices and anchor damage		4, 5	5

North Atlantic				
	KNOWN THREATS	EXTENT	LIFE STAGE AFFECTED	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR B: Overutilization	Historic-intentional harvest	1	1, 2, 4, 5	4
	Current- intentional harvest	3	1, 2, 4, 5	1, (possibly 4 for Nicaragua)
FACTOR C: Disease	FP	1	4, 5	3 (new FP in TX)
FACTOR C: Predation	Beach and water	1	1, 2, 3, 4	6
FACTOR D: Inadequacy of Regulation	Current international	MARPOL-implementation and enforcement	3, 4, 5	1
	Country	Cayman- size limit- between 40 and 6. Haiti- regulations ignored, Nicaragua- consuming turtle eggs prohibited but continues, Panama- egg use and harvest	4	1
	Local	FL- CCL control line- armoring continues. Lighting and beach furniture- depend on funding for compliance and commitment by County or Municipality	1, 2, 3	1

North Atlantic				
	KNOWN THREATS	EXTENT	LIFE STAGE AFFECTED	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR E: Other	Incidental Bycatch in Fishing Gear	1	4, 5	5,6 (TED)
	Vessel Strikes	2	4, 5	5
	Climate Change	1	1, 2, 3, 4, 5	3
	Natural Disasters	1	1, 2, 4	3
	Contaminants	1	3, 4, 5	3
	Dredging	2	4, 5	2

Mediterranean				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal Development with lighting	1	1, 2, 3	2
	Erosion from storm events and sand extraction and existing jetty	2	1, 2	2
	Marine Pollution	1	1, 4,5	2
	Climate change: Sea level rise and increased storm events- loss of habitat	1	1, 2, 3, 4, 5	3
	Beach Driving	2	1, 2, 3	2
	Human activity	2	1, 2, 3,	2
	Trawling	1	4, 5	2
	Fishing practices	2	4, 5	2
FACTOR B: Overutilization	Historic-intentional harvest	1	1, 2, 4, 5	4
	Current-intentional harvest	3	1, 2, 4, 5	1, (Egypt)
FACTOR C: Disease	FP	4	0	0
FACTOR C: Predation	Beach	1	1, 2, 3	6

Mediterranean				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR D: Inadequacy of Regulation	Current international- all countries participate, some, none			
	Country			
	Local			
FACTOR E: Other	Incidental Bycatch in Fishing Gear	1	4, 5	2
	Vessel Strikes	2	4	3
	Power Generation Activity	2	4, 5	3
	Pollution	1	3, 4	2
	Climate Change	1	2, 3, 4, 5	3
	Natural Disasters	1	1, 2, 3	3

South Atlantic				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal Development with lighting including armoring and jetties	2	1, 2, 3	3
	Erosion from storm events and sand extraction	1	1, 2	3
	Beach engineering	2	1, 2	2,6
	Climate change: Sea level rise and increased storm events-loss of habitat	1	1, 2	3
	Beach and pedestrian traffic	3	1, 2	2
	Beach and marine pollution (runoff and sedimentation	2	1, 2, 3, 4, 5	5
	Non-Native vegetation	2	1, 2	1
	Fishing practices	2	4, 5	5
	Dredging	2	4, 5	2
FACTOR B: Overutilization	Historic-intentional harvest	1	1, 2, 4, 5	4
	Current -intentional harvest	2	1, 2, 4, 5	1, (Northeast Brazil, Suriname, Bioko, Guinea-Bissau, Orange National Park)
FACTOR C: Disease	FP	1	4, 5	3
FACTOR C: Predation	Beach and water	1	1, 2, 3, 4, 5	6

South Atlantic				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
	Current international- all countries participate, some, none		4, 5	1
FACTOR D: Inadequacy of Regulation	Country	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	1, 2, 3, 4, 5	1
	Incidental Bycatch in Fishing Gear	1	4, 5	5,6 (TED)
	Pollution and Oil Exploration	1	3, 4, 5	3
	Climate Change	1	1, 2, 3, 4, 5	3
FACTOR E: Other	Natural Disasters	2	1, 2, 4	3

Southwest Indian				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal Development with lighting	2	1, 2, 3	3 (lighting being addressed on Aldabra)
	Erosion from storm events and sand extraction	1	1, 2	2
	Climate change: Sea level rise and increased storm events-loss of habitat	1	1, 2	3
	Dredging	2	4, 5	2
FACTOR B: Overutilization	Historic-intentional harvest	1	1, 2, 4, 5	4
	Current- intentional harvest	2	1, 2, 4, 5	1, (Maldives, Mahe, Praslin, La Digue , Eparses Islands)
FACTOR C: Disease	FP	2	4, 5	5
FACTOR C: Predation	Beach	1	2, 3	5
FACTOR D: Inadequacy of Regulation	Current international		4, 5	1
	Country			
	Local			
FACTOR E: Other	Incidental Bycatch in Fishing Gear	1	4, 5	5
	Climate Change	1	1, 2, 3, 4, 5	3
	Natural Disasters	2	1, 2, 4	3

North Indian				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal Development with lighting	2	1, 2, 3	3
	Erosion from storm events and sand extraction	1	1, 2	2
	Vehicles and boats on beach	2 (Oman)	1, 2, 3	
	Beach and marine pollution	2?	?	
	Climate change: trophic changes to foraging	1	1, 2	3
	Fishing practice-trawling	1		
	Dredging	2		
FACTOR B: Overutilization	Historic-intentional harvest	1	1, 2, 4, 5	4
	Current- intentional harvest	2	1, 2, 4, 5	1 (Oman, Saudi Arabia, Qatar, Yemen, India, Eritrea, Iran, and Sri Lanka)
FACTOR C: Disease	FP	Not known		
FACTOR C: Predation	Beach	1	2, 3	5

North Indian				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR D: Inadequacy of Regulation	Current international		4, 5	1
	Country		Djibouti- only recently more active; Somalia- insufficient and not enforced; Sudan- fishing law but not specific protection for sea turtles; Yemen- enforcement undefined	
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

East Indian-West Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal Development with lighting (oil flares off shore)	1	1, 2, 3	3
	Erosion from storm events (tsunami) and sand mining and ports	2	1, 2	2
	Vehicle and pedestrian traffic	2	2	2
	Climate change: sea level rise	1	1, 2	3
	Pollution: siltation and degradation	1	3, 4, 5	5
	Fishing practices including seagrass collection	2	3, 4, 5	5
FACTOR B: Overutilization	Historic-intentional harvest	1	1, 2, 4, 5	4
	Current- intentional harvest	2	1, 2, 4, 5	5
FACTOR C: Disease and Predation	Disease (FP)	1	4, 5	5

East Indian-West Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR D: Inadequacy of Regulation	Current international	1	4, 5	1
	Country		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	5
FACTOR E: Other	Incidental Bycatch in Fishing Gear	1	4, 5	5, 6 (TEDs in some areas)
	Pollution and debris	3	4, 5	3
	Climate Change	1	1, 2, 3, 4, 5	3
	Natural Disasters	2	1, 2, 4	3

Central West Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal Development (armoring) with lighting	1	1, 2, 3	3
	Erosion from storm events and sand extraction	1	1, 2	3
	Vehicle and pedestrian traffic	3	1, 2, 3	3
	Non-native vegetation	3	1, 2	5
	Pollution: beach and sedimentation and runoff	2	1, 2, 3, 4, 5	5
	Fishing practices (destructive)	2	3, 4, 5	5
	Dredging	2	4, 5	5
FACTOR B: Overutilization	Historic-intentional harvest	1	1, 2, 4, 5	4
	Current- intentional harvest	1	1, 2, 4, 5	4
FACTOR C: Disease	FP	2	4, 5	5
FACTOR C: Predation	Beach	1	2, 3	5
FACTOR D: Inadequacy of Regulation	Current international- all countries participate, some, none	not all	4, 5	1
	Country		Micronesia- local consumption, Palau- only certain life stages	5

Central West Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR E: Other	Incidental Bycatch in Fishing Gear	1	4, 5	5
	Pollution	3	4, 5	3
	Vessel strikes	3	4, 5	5
	Climate Change	1	1, 2, 3, 4, 5	3
	Natural Disasters	2	1, 2, 4	3

Southwest Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal Development (armoring/ 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
FACTOR C: Predation	Beach and water	1	1, 2, 3, 4	5

Southwest Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR D: Inadequacy of Regulation	Current international		4, 5	1
	Country	New Caledonia- take prohibited only during certain time period		5
FACTOR E: Other	Incidental Bycatch in Fishing Gear	1	4, 5	5, 6 (TEDs in Australia)
	Shark control programs	2	4, 5	5,6
	Vessel strikes, port dredging, military activities	2	4, 5	3
	Toxic compounds and debris	2	4, 5	5
	Climate Change- sea surface temp	1	1, 2, 3, 4, 5	5
	Natural Disasters	2	1, 2	5

Central South Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal development and associated lighting	3	1, 2, 3	3
	Erosion from sand mining	1	1, 2	3
	Pollution: beach and sedimentation and runoff	2	1, 2, 3, 4, 5	5
	climate change: sea level	2	1, 2	3
	Dredging	2	4, 5	5
FACTOR B: Overutilization	Historic-intentional harvest	2	1, 2, 4, 5	4
	Current- intentional harvest	1	1, 2, 4, 5	2
FACTOR C: Disease	FP	0	0	0
FACTOR C: Predation	Beach	1	1, 2, 3	5, 6

Central South Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR D: Inadequacy of Regulation	Current international		4, 5	1
	Country	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
FACTOR E: Other	Incidental Bycatch in Fishing Gear	1	4, 5	5, 6 (TEDs in Australia)
	Marine debris and pollution	2	3, 4, 5	5
	Climate change -	2	4, 5	5
	Natural Disasters	2	1, 2	5

Central North Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal development and associated lighting	3	1, 2	5
	Armament and erosion	2	1, 2	3
	Sea wall	2 portions of FFS	1, 2	1
	Beach driving	3	1, 2	2
	Pollution: beach and sedimentation and runoff, contaminants, vessel grounds	3	1, 2, 3, 4, 5	5
	climate change: sea level and storm events , trophic changes	1	1, 2	3
FACTOR B: Overutilization	Historic-intentional harvest	3	1, 2, 4, 5	4
	Current- intentional harvest	3	1, 2, 4, 5	1
FACTOR C: Disease	FP	1	4,5	2
FACTOR C: Predation	Beach	3	2, 3	1
FACTOR D: Inadequacy of Regulation	Current international- all countries participate, some, none			
	Country			5

Central North Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR E: Other	Incidental Bycatch in Fishing Gear	1	4, 5	5
	Pollution-red tide	1	4, 5	5
	Boat strikes	2	4, 5	5
	Climate change	1	1, 2, 3, 4, 5	3

East Pacific				
	KNOWN THREATS	EXTENT	LIFE STAGE	LEVEL
		1. Throughout DPS, 2. Portion of DPS on high density nesting beaches and/or protected beaches, 3. Portion of DPS on low density nesting beaches	1. Nesting female on beach, 2. Egg, 3. Hatchling, 4. Juvenile (neritic and oceanic), 5. Adult (neritic and oceanic)	1. Present, 2. Consistent, 3. Increasing, 4. Threat to Population Stability, 5. Unknown if Increasing or Decreasing, 6. Regular Conservation Practice Minimizes in Some or All of DPS
FACTOR A: Habitat	Coastal development and associated lighting	3	1, 2	5
	Foot traffic	3	1, 2	5
	Pollution: beach and sedimentation and runoff, contaminants, vessel	3	4, 5	5
	climate change: sea level and storm events , trophic changes	1	1, 2	3
FACTOR B: Overutilization	Historic-intentional harvest	1	1, 2, 4, 5	4
	Current- intentional harvest	1	1, 2, 4, 5	1
FACTOR C: Disease	FP	1	4, 5	3
FACTOR C: Predation	Beach and water	1	1, 2, 3, 4, 5	5, 6
FACTOR D: Inadequacy of Regulation	Current international- all countries participate, some, none			
	Country			
	Local			
FACTOR E: Other	Incidental Bycatch in Fishing Gear	1	4, 5	5, 6 (TEDs)
	Marine debris and pollution	2	3, 4, 5	5
	Vessel Interaction	2	4, 5	5
	Climate change			

Conservation Efforts to protect all life stages of green turtles are affected 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.

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

DPS	#1	#2	#3	#4	#5	#6	#7	#8	#9	#10	#11
	North Atlantic	**Mediterranean**	**South Atlantic**	**Southwest Indian**	**North Indian**	**East Indian-West Pacific**	**Central West Pacific**	**Southwest Pacific**	**Central South Pacific**	**Central North Pacific**	**East Pacific**
Convention on International Trade in Endangered Species of Wild Fauna and Flora	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Convention on the Conservation of European Wildlife and Natural Habitats		✓									
Convention for the Co-operation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region (Abidjan Convention)			✓								
Convention for the Protection of the Marine Environment of the North-East Atlantic	✓										
Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region, Protocol Concerning Specially Protected Areas and Wildlife (SPAW)	✓		✓								
Convention for the Protection of the Marine Environment and Coastal Area of the South-East Pacific (Lima Convention)											✓
Convention for the Protection of Natural Resources and Environment of the South Pacific Region						✓	✓		✓		

DPS	#1	#2	#3	#4	#5	#6	#7	#8	#9	#10	#11
	North Atlantic	**Mediterranean**	**South Atlantic**	**Southwest Indian**	**North Indian**	**East Indian-West Pacific**	**Central West Pacific**	**Southwest Pacific**	**Central South Pacific**	**Central North Pacific**	**East Pacific**
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 for the Prohibition of Fishing with Long Drift nets in the South Pacific							✓				
Council Regulation (EC) No. 1239/98 of 8 June 1998 Amending Regulation (EC) No. 894/97 Laying Down Certain Technical Measures for the Conservation of Fishery Measures (Council of the European Union)			✓								
Food and Agriculture Organization Technical Consultation on Sea Turtle-Fishery Interactions	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
Forum Fisheries Authority (FFA)								✓	✓		
Indian Ocean Tuna Commission (IOTC)				✓	✓	✓		✓			
Indian Ocean Southeast Asian Marine Turtle Memorandum of Understanding (IOSEA)				✓	✓	✓	✓	✓			

DPS	#1	#2	#3	#4	#5	#6	#7	#8	#9	#10	#11
	North Atlantic	**Mediterranean**	**South Atlantic**	**Southwest Indian**	**North Indian**	**East Indian-West Pacific**	**Central West Pacific**	**Southwest Pacific**	**Central South Pacific**	**Central North Pacific**	**East Pacific**
Inter-American Convention for the Protection and Conservation of Sea Turtles	✓		✓	✓					✓	✓	✓
International Convention for the Presentation of Pollution from Ships (MARPOL)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	
International Union for Conservation of Nature (IUCN)	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓
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).	✓	✓	✓								

DPS	#1	#2	#3	#4	#5	#6	#7	#8	#9	#10	#11
	North Atlantic	**Mediterranean**	**South Atlantic**	**Southwest Indian**	**North Indian**	**East Indian-West Pacific**	**Central West Pacific**	**Southwest Pacific**	**Central South Pacific**	**Central North Pacific**	**East Pacific**
Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region				✓				✓			
Protocol Concerning Specially Protected Areas and Biological Diversity in the Mediterranean		✓									
Ramsar Convention on Wetlands		✓	✓		✓	✓	✓	✓	✓	✓	
Secretariat of the Pacific Regional Environment Programmed (SPREP)						✓	✓	✓	✓	✓	
South-East Atlantic Fisheries Organization (SEAFO)			✓								
Torres Strait Treaty of 1978								✓			
United Nations Convention on the Law of the Sea (UNCLOS)	✓	✓	✓		✓	✓	✓	✓	✓	✓	
United Nations Resolution 44/225 on Large-Scale Pelagic Drift net Fishing	✓	✓	✓		✓	✓	✓	✓	✓	✓	
United States Magnuson-Stevens Fishery Conservation and Management Act	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓	✓

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.

	Element 1	Element 2	Element 3	Element 4	Element 5	Element 6
	Abundance	Trends / Productivity	Spatial Structure	Diversity / Resilience	Five-Factor Analysis	Conservation Efforts
1. North Atlantic	1.18	1.18	1.45	1.36	-0.45	0.82
2. Mediterranean	3.92	2.75	3.58	3.08	-1.25	0.50
3. South Atlantic	1.58	1.92	1.33	1.67	-0.83	0.75
4. Southwest Indian	1.25	1.75	1.42	1.58	-0.75	0.75
5. North Indian	1.42	2.00	1.58	2.00	-0.92	0.33
6. East Indian-West Pacific	1.67	3.08	1.67	1.50	-1.50	0.5
7. Central West Pacific	2.50	2.42	2.17	2.17	-1.08	0.67
8. Southwest Pacific	1.17	1.67	1.50	1.42	-0.67	0.58
9. Central Soutth Pacific	3.18	2.91	1.91	2.18	-1.27	0.55
10. Central North Pacific	2.67	1.33	3.00	2.58	-0.92	0.50
11. East Pacific	1.64	1.18	1.73	1.64	-0.91	1.09

The SRT then assessed the probability that each DPS will reach a quasi-extinction 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.

	DPS 1	DPS 2	DPS 3	DPS 4	DPS 5	DPS 6	DPS 7	DPS 8	DPS 9	DPS 10	DPS 11
	N Atlantic	Mediterranean	S Atlantic	SW Indian	North Indian	East Indian/ West Pacific	CW Pacific	SW Pacific	CS Pacific	CN Pacific	E Pacific
<1%	**87.0**	10.1	**69.0**	**71.3**	**68.4**	**47.5**	**43.3**	**72.5**	**38.2**	**47.8**	**63.6**
1-5%	3.0	11.8	16.5	13.6	19.8	17.3	19.3	9.1	19.9	15.9	16.7
6-10%	1.4	17.6	9.9	7.6	9.7	11.2	13.8	10.6	20.5	14.1	6.4
11-20%	4.1	**27.9**	4.2	5.5	1.7	9.4	12.5	5.4	8.2	7.5	7.8
21-50%	4.1	23.9	0.4	2.0	0.5	7.1	8.6	2.4	9.5	9.8	5.0
>50%	0.5	8.8	0.0	0.0	0.0	7.5	2.7	0.0	3.7	4.9	0.5
at least 0%	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0	100.0
at least 1%	13.0	89.9	31.0	28.7	31.6	52.5	56.8	27.5	61.8	52.2	36.4
at least 6%	10.0	78.2	14.5	15.1	11.8	35.2	37.5	18.4	41.9	36.3	19.6
at least 11%	8.6	60.6	4.6	7.5	2.2	24.0	23.8	7.8	21.5	22.2	13.3
at least 21%	4.5	32.7	0.4	2.0	0.5	14.6	11.3	2.4	13.3	14.7	5.5
at least 50%	0.5	8.8	0.0	0.0	0.0	7.5	2.7	0.0	3.7	4.9	0.5

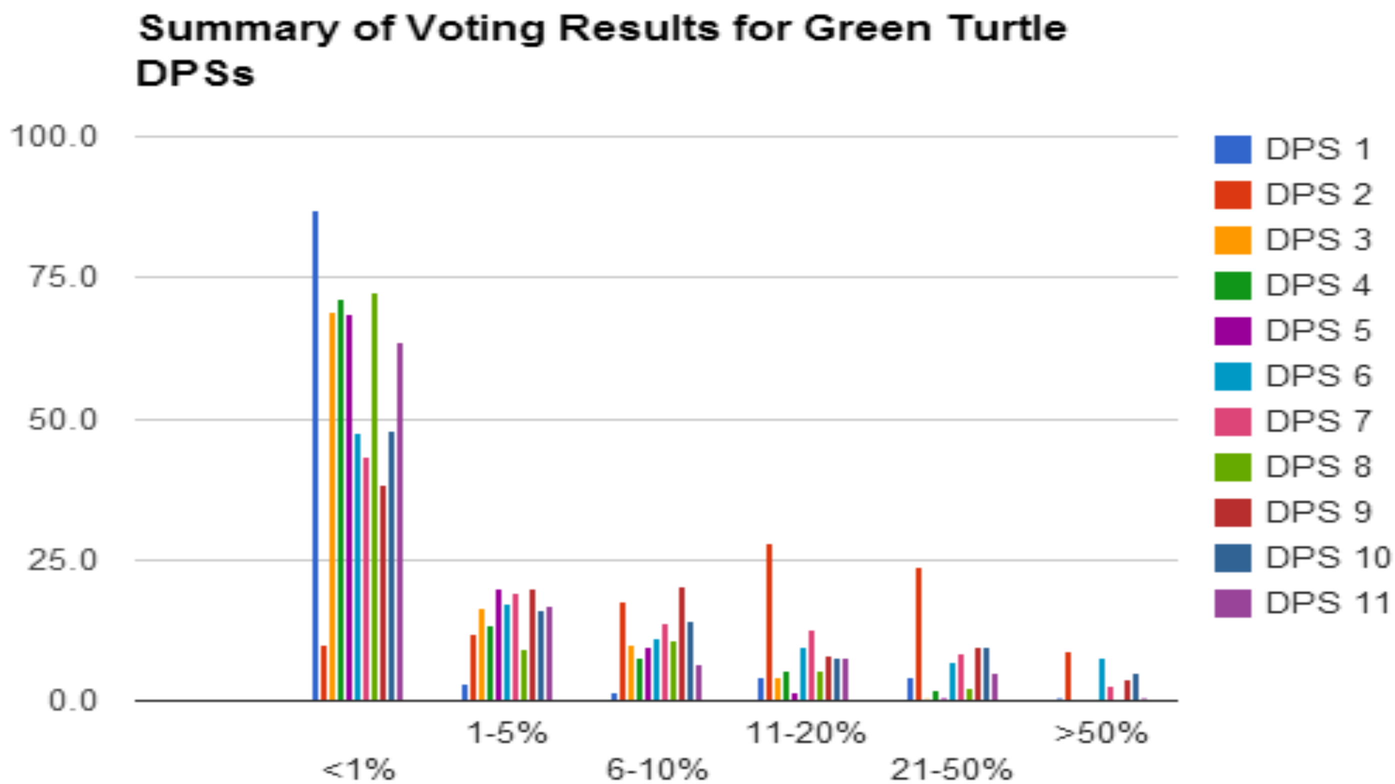

Figure 16.1. Bar graph of risk threshold scores.

17. LITERATURE CITED

ABC, 2010. Timeline: 20 years of major oil spills. http//www.abc.net.au/news/2010-05-03/timeline-20-years-of-major-oil-spills/419898, Accessed 6 May 2013.

Abdulqader, E.A.A., 2008. Marine turtle mortality in the waters of the Kingdom of Bahrain: Report of the Dead Marine Turtle Monitoring Program for the last quarter of 2007. Bahrain Center for Studies and Research. pp 37.

Abdulqader, E.A.A., 2010. Turtle captures in shrimp trawls in Bahrain 13, 307–318.

Abdulqader, E.A.A., Miller, J., 2012. Marine turtle mortalities in Bahrain territorial waters. Chelonian Conserv. Biol. 11, 133–138.

Abe, O., Shibuno, T., Takada, Y., Hashimoto, K., Tanizaki, S., Hisakazu, I., Funakura, Y., Sano, K., Okamura, Y., 2003. Nesting populations of sea turtle in Ishigaki Island, Okinawa, in: Proceedings of the 4th SEASTAR2000 Workshop, 11–13 December 2003. Bangkok, pp. 40–43.

Abe, O., Takada, Y., Shibuno, T., Hashimoto, K., Tanizaki, S., 1998. Nesting activity of sea turtles on Ishigaki Island, Ryukyu Islands, Japan, in: Epperly, S.P., Braun, J. (Eds.), Seventeenth Annual Sea Turtle Symposium. Orlando, Florida, p. 115.

Ackerman, R.A., 1997. The nest environment and the embryonic development of sea turtles, in: The Biology of Sea Turtles. CRC Press, Boca Raton, Florida, pp. 83–106.

Adnyana, I.B.W., Ladds, P.W., Blair, D., 1997. Observations of fibropapillomatosis in green turtles (*Chelonia mydas*) in Indonesia. Aust. Vet. J. 10, 737–742.

Adnyana, I.B.W., Soede, L.P., Gearheart, G., Halim, M.H., 2008. Status of green turtle (*Chelonia mydas*) nesting and foraging populations of Berau, East Kalimantan, Indonesia, including results from tagging and telemetry. Indian Ocean Turt. Newsl. 7, 2–11.

Aeby, G., Aletto, S.C., Anderson, P., Carroll, B., DiDonato, E., DiDonato, G., Farmer, V., Fenner, D., Gove, J., Gulick, S., Houk, P., Lundblad, E., Nadon, M.O., Riolo, F., Sabater, M., Schroeder, R., Speicher, M., Tuitele, C., Tagarino, A.P., Vaitautolu, S., Vaoli, E., Vargas-Angel, B., Vroom, P.S., 2008. The State of Coral Reef Ecosystems of American Samoa, in: Waddell, J., Clarke, A.M. (Eds.), The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States: 2008. NOAA Technical Memorandum NOS NCCOS 73. NOAA/NCCOS Center for Coastal Monitoring and Assessment's Biogeography Team. Silver Spring, MD, pp. 307–351.

Agardy, T., Gil Hernandez, R.A., 1989. Biology and conservation of sea turtles nesting in Quintana Roo, Mexico, during 1988, in: Eckert, S. A., Eckert, K. L. & Richardson, J.H. (Compilers). (Ed.), Proceedings of the Ninth Annual Workshop on Sea Turtle Conservation and Biology. National Technical Information Service, NOAA Tech. Memo. NMFS-SEFC 232., Springfield, VA, pp. 199–200.

Aguirre, A.A., Lutz, P.L., 2004. Marine turtles as sentinels of ecosystem health: Is fibropapillomatosis an indicator? Ecohealth 1, 275–283.

Ahamada, S., 2008. Estimation and conservation of the sea turtle population in Moheli, Comoros, in: Rees, A.F., Frick, M., Panagopoulou, A., Williams, K. (Eds.), Proceedings of the Twenty-Seventh Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-569, pp. 70–71.

Ahmed, R., Khan, A., Nishat, A., 2006. Community-based awareness programme for the conservation of sea turtles in Bangladesh, in: Pilcher, N.J. (Ed.), Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-536, p. 25.

Ai, L.X., 2001. Sea Turtle Conservation and Research in Con Dao Islands, in: Proceedings of the Training Workshop on Marine Turtle Research and Conservation in Viet Nam. IUCN, Hanoi, pp. 147–148.

Aiken, J.J., Godley, B.J., Broderick, A.C., Austin, T., Ebanks-petrie, G., Hays, G.C., 2001. Two hundred years after a commercial marine turtle fishery: the current status of marine turtles nesting in the Cayman Islands. Oryx 35, 145–151.

Akcinar, S.C., Guclu, O., Taskavak, E., Türkozan, O., 2006. Golden jackal predation on marine turtles in Goksu Delta, Turkey, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Proceedings of the Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece., p. 120.

Al Suweidi, A.S., Wilson, K.D.P., Healy, T., Vanneyre, L., 2012. First contemporary record of green turtle (*Chelonia mydas*) nesting in the United Arab Emirates. Mar. Turt. Newsl. 133, 16–17.

Al-Abdessalaam, T.Z., Das, H.S., Kabbara, L., 2008. Sea turtles of the United Arab Emirates. Indian Ocean - South-East Asian Mar. Turt. Memo. Underst. Profile Mon.

Alefaio, S.T., Alefaio, A.R., 2006. Turtle monitoring on Funafuti, Tuvalu, 4th December-14th December. Administered by the Institute of Marine Resoruces, the Universtiy of the South Pacific, Suva. 8 pages.

Alfaro-Shigueto, J., Mangel, J.C., Pajuelo, P., Caceres, C., Seminoff, J.A., Dutton, P.H., 2006. Bycatch in Peruvian artisanal fisheries: gillnets versus longlines, in: Proceedings of the Twenty-Seventh Annual Symposium on Sea Turtle Biology and Conservation.

Alfaro-Shigueto, J., Mangel, J.C., Seminoff, J.A., Dutton, P.H., 2008. Demography of loggerhead turtles *Caretta caretta* in the southeastern Pacific Ocean: fisheries-based observations and implications for management. Endanger. Species Res. 5, 129–135.

Alhafez, A.A., 2010. Volunteerism in action: marine turtle conservation in Kuwait [WWW Document]. Indian Ocean - South-East Asian Mar. Turt. Memo. Underst. Profile Mon. URL http://www.ioseaturtles.org/pom_detail.php?id=104

Ali, A., Ibrahim, K., 2000. Crab predation on green turtle (*Chelonia mydas*) eggs incubated on a natrual beach and in turtle hatcheries, in: Proceedings of the 3rd Workshop on SEASTAR 2000. Graduate School of Informatics, Kyoto University, pp. 95–100.

AlKindi, A.Y.A., Mahmoud, I.Y., Al-Gheilani, H.M., Bakheit, C.S., Al-Habsi, A.A., Al-Kiyumi, A.A., 2003. Comparative study of the nesting behavior of the green turtle, *Chelonia mydas*, during high-and low-density nesting periods at Ras Al-Hadd Reserve, Oman. Chelonian Conserv. Biol. 4, 603–611.

AlKindi, A.Y.A., Mahmoud, I.Y., Al-Kiyumi, A.A., Al-Bahry, S.N., Elshafie, A., Al-Siyabi, S.S., 2008. The status of the nesting population of the green turtle (*Chelonia mydas*) during low and high nesting periods in 2006 and 2007 with reference to cyclone Gonu in Ras al-Haad, Oman, in: Dean, K., López-Castro, M.C. (Eds.), Proceedings of the Twenty-Eighth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NOAA NMFS-SEFSC-602, p. 158.

Allard, M.W., Miyamoto, M.M., Bjorndal, K.A., Bolten, A.B., Bowen, B.W., 1994. Support for Natal Homing in Green Turtles from Mitochondrial DNA Sequences. Copeia 1994, 34–41.

Allen, C.R., Forys, E.A., Rice, K.G., Wojcik, D.P., 2001. Effects of fire ants (Hymenoptera: Formicidae) on hatching turtles and prevalence of fire ants on sea turtle nesting beaches in Florida. Florida Entomol. 84, 250–253.

Allen, M.S., 2007. Three millennia of human and sea turtle interactions in Remote Oceania. Coral Reefs 26, 959–970.

Almeida, A.P., Moreira, L.M.P., Bruno, S.C., Thomé, J.C.A., Martins, A.S., Bolten, A.B., Bjorndal, K.A., 2011. Green turtle nesting on Trindade Island, Brazil: abundance, trends, and biometrics. Endanger. Species Res. 14, 193–201.

Al-Merghani, M., Miller, J.D., Al-Mansi, A., Khushaim, O., Pilcher, N.J., 1996. The marine turtles of the Arabian Gulf, in: Krupp, F., Abuzinada, A.H., Nader, I.A. (Eds.), A Marine Wildlife Sanctuary for the Arabian Gulf, Environmental Research and Conservation Following the 1991 Gulf War Oil Spill. National Commission for Wildlife Conservation and Development, Riyadh and Senckenbergische Naturforschende Gesellschaft, Frankfurt, Germany, pp. 351–359.

Al-Merghani, M., Miller, J.D., Pilcher, N.J., Al-Mansi, A.M.A., 2000. The green and hawksbill turtles in the Kingdom of Saudi Arabia: synopsis of nesting studies 1986-1997. Fauna Arab. 18, 369–384.

Al-Muraikhi, N., Al-Jabri, A., Pilcher, N.J., 2005. The status of sea turtles in Qatar. Indian Ocean - South-East Asian Mar. Turt. Memo. Underst. Profile Mon.

Alpers, W., 1993. Potential applications of remote sensing techniques for monitoring marine pollution events in the Mediterranean Sea, in: STOA Project on Pollution of the Mediterranean Sea, Directorate General for Research, 10-11 September 1993.

Al-Saady, S.M., Al-Kiyumi, A.A., Mendonça, V.M., 2005. Oman sea turtles migration routes – evidence from returned tags, in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Tech Memo NMFS-SEFSC 528, Philadelphia, PA, USA, p. 91.

Alvarado-Díaz, J., Arias-Coyotl, E., Delgado-Trejo, C., 2003. Clutch Frequency of the Michoacán Green Seaturtle. J. Herpetol. 37, 183–185.

Alvarado-Díaz, J., Delgado-Trejo, C., Suazo-Ortuño, I., 2001. Evaluation of the Black Turtle Project in Michoacán, México. Mar. Turt. Newsl. 92, 4–7.

Alvarado-Díaz, J., Figueroa, A., 1990. The ecological recovery of sea turtles of Michoacan, Mexico: Special attention: black turtle (Chelonia agassizii). Final Report for USFWS and WWF-US.

Alvarado-Díaz, J., Figueroa, A., 1992. Recapturas post-anidatorias de hembras de tortuga marina negra (Chelonia agassizii) marcadas en Michoacán, México. Biotropica 24, 560–566.

Alvez, R.R.., Rosa, I.L., 2006. Fron cnidarians to mammals: The use of animals as remedies in fishing communities in NE Brazil. J. Ethnopharmacol. 107, 259–276.

Amerson, A.B., 1971. The natural history of French Frigate Shoals, northwestern Hawaiian islands. Atoll Res. Bull. 150, 400 pages.

Amerson, A.B., Clapp, R.B., Wirtz, W.O., 1974. The Natural History of Pearl and Hermes Reef, Nortwestern Hawaiian Islands. Atoll Res. Bull. 174, 347 pages.

Amerson, A.B., Whistler, W.A., Schwaner, T.D., 1982a. Wildlife and wildlife habitat of American Samoa. I. Accounts of flora and fauna. United States Department of the Interior, Fish and Wildlife Service, Washington, D.C.

Amerson, A.B., Whistler, W.A., Schwaner, T.D., 1982b. Wildlife and wildlife habitat of American Samoa. II. Accounts of flora and fauna. United States Department of the Interior, Fish and Wildlife Service, Washington, D.C.

Amorocho, D.F., Abreu-Grobois, F.A., Dutton, P.H., Reina, R.D., 2012. Multiple Distant Origins for Green Sea Turtles Aggregating off Gorgona Island in the Colombian Eastern Pacific. PLoS One 7, e31486.

Amorocho, D.F., Reina, R., 2007. Feeding ecology of the East Pacific green sea turtle*Chelonia mydas* agassiziiat Gorgona National Park, Colombia. Endanger. Species Res. 3, 43–51.

Amorocho, D.F., Reina, R., 2008. Intake passage time, digesta composition and digestibility in East Pacific green turtles (*Chelonia mydas* agassizii) at Gorgona National Park, Colombian Pacific. J. Exp. Mar. Bio. Ecol. 360, 117–124.

Anan, Y., Kunito, T., Watanabe, I., Sakai, H., Tanabe, S., 2001. Trace element accumulation in hawksbill turtles (*Eretmochelys imbricata*) and green turtles (*Chelonia mydas*) from Yaeyama Islands, Japan. Environ. Toxicol. Chem. 20, 2802–2814.

Andrews, H. V., 2000. Current marine turtle situation in the Andaman and Nicobar Islands - An urgent need for conservation action. Kachhapa 3, 21–25.

Andrews, H. V., Krishnan, S., Biswas, P., 2001. The status and distribution of marine turtles around the Andaman and Nicobar Archipelago. Survey Report of Centre for Herpetology, Madras Crocodile Bank Trust. 29 pp.

Andrews, H. V., Krishnan, S., Biswas, P., 2006a. Distribution and status of marine turtles in the Andaman and Nicobar Islands, in: Shanker, K., Choudhury, B.C. (Eds.), Marine Turtles of the Indian Subcontinent, Part 2: Marine Turtles of the Bay of Bengal. Universities Press, Hyderabad, Hyderabad, India, pp. 33–57.

Andrews, H. V., Tripathy, A., Aghue, S., Glen, S., John, S., Naveen, K., 2006b. The status of sea turtle populations in the Andaman and Nicobar Islands, in: Shanker, K., Andrews, H.V. (Eds.), Towards an Integrated and Collaborative Sea Turtle Conservation Programme in India: A UNEP/CMS-IOSEA Project Report. Centre for Herpetology/Madras Crocodile Bank Trust, Post Bag 4, Mamallapuram, Tamil Nadu, India.

Angliss, R.P., Silber, G.K., Merrick, R., 2002. Report of a workshop on developing recovery criteria for large whale species. NOAA Tech. Memo. NMFS-OPR-21 32.

Arana, G.L.E., 2006. Legislation that protects sea turtles in Guatemala, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece, p. 121.

Arenas, A., Herrera, R., Calderón, C., Gómez, L., Raigoza, R., 2007. Sea turtle protection and conservation program at Quintana Roo, México, in: Mast, R.B., Hutchinson, B.J., Hutchinson, A.H. (Eds.), Proceedings of the Twenty-Fourth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-567, San Jose, Costa Rica, p. 112.

Argument, D., MacKay, K.T., Krueger, B.H., 2009. Foraging turtles around Tetepare Island, Solomon Islands. Mar. Turt. Newsl. 123, 18–20 ST – Foraging turtles around Tetepare Islan.

Aridjis, H., 1990. Mexico proclaims total ban on harvest of turtles and eggs. Mar. Turt. Newsl. 50, 1–3.

Arinal, I., 1997. Marine turtle management in Meru Betiri National Park, in: Noor, Y.R., Lubis, I.R., Ounsted, R., Troeng, S., Abdullah, A. (Eds.), Proceedings of the Workshop on Marine Turtle Research and Management in Indonesia. Wetlands International/PHPA/Environment Australia.

Arthur, K.E., Boyle, M.C., Limpus, C.J., 2008. Ontogenetic changes in diet and habitat use in green sea turtle (*Chelonia mydas*) life history. Mar. Ecol. Prog. Ser. 362, 303–311.

Arthur, R., Kelkar, N., Alcoverro, T., Madhusudan, M.D., 2013. Complex ecological pathways underlie perceptions of conflict between green turtles and fishers in the Lakshadweep Islands. Biol. Conserv. 167, 25–34.

Arvy, C., Maigret, J., Tijane, D.A., Colas, F., 1996. Observations of carapaces of sea turtles on Mauritanian coast (Cap Timirisit[Timirist]-Senegal frontier). Bull. La Soc. Herpetol. Fr. 1997, 79 3.

Asrar, F.F., 1999. Decline of marine turtle nesting populations in Pakistan. Mar. Turt. Newsl. 13–14.

Atkinson, C., Nolasco del Aguila, D., Harrison, E., 2011. Report on the 2010 green turtle program at Tortuguero, Costa Rica. Sea Turtle Conservancy and the Ministry of Envirionment, Engery and Telecommunications of Costa rica.

Atkinson, C., Ramos, L.B., Harrison, E., 2010. Report on the 2009 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation and the Ministry of Envirionment, Engery and Telecommunications of Costa rica.

Aureggi, M., 2001. Green turtle monitoring programme: Kazanli Beach, Turkey, 2001. UNEP, Mediterranean Action Plan, Regional Activity Centre for Specially Protected Areas – Boulevard de l'Environnement, BP 337 – 1080 Cedex, Tunisia.

Aureggi, M., Adulyanukosol, K., 2006. Marine turtle conservation in Thailand: A post tsunami recovery, in: Book of Abstracts of the Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece.

Aureggi, M., Gerosa, G., Chantrapornsyl, S., 2004. An update of sea turtle nesting along the Andaman coast of Thailand: 1996-2000, in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings 21st International Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-528, pp. 98–100.

Aureggi, M., Gerosa, G., Yerli, S. V., 1999. Observations on predation of marine turtle nests at Akyatan, Turkey, eastern Mediterranean. Chelonian Conserv. Biol. 3, 487–489.

Aureggi, M., Khalil, M., 2010. Lebanon, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Priorities. IUCN, Gland, Switzerland, pp. 149–156.

Aureggi, M., Rizk, C., Venizelos, L., 2005. Survey on sea turtle nesting activity South Lebanon, 2004.

Australian Government Department of Sustainability, Environment, Water, Population and Communities Species Profile and Threats Database. *Chelonia mydas* - green turtle [WWW Document], 2012. URL http://www.environment.gov.au/cgi-bin/sprat/public/publicspecies.pl?taxon_id=1765

Autar, L., 1994. Sea turtles attacked and killed by jaguars in Suriname. Mar. Turt. Newsl. 67, 11–12.

Avens, L., Goshe, L.R., Harms, C.A., Anderson, E.T., Goodman Hall, A., Cluse, W.M., Godfrey, M.H., Braun-McNeill, J., Stacy, B., Bailey, R., Lamont, M.M., 2012. Population characteristics, age structure, and growth dynamics of neritic juvenile green turtles in the northeastern Gulf of Mexico. Mar. Ecol. Prog. Ser. 458, 213–229.

Avens, L., Snover, M.L., 2013. Age and age esimtation in sea turtles, in: Wyneken, J., Lohmann, K.J., Musick, J.A. (Eds.), The Biology of Sea Turtles Volume III. CRC Press Boca Raton, FL, pp. 97–133.

Awbrey, F.T., Leatherwood, S., Mitchell, E.D., Rogers, W., 1984. Nesting green sea turtles (*Chelonia mydas*) on Isla Clarion, Islas Revillagigedos, Mexico. Bull. South. Calif. Acad. Sci. 83, 69–75.

Azanza, J., Moncada, F.G., Gerhartz, J., Forneiro, Y., Medina, Y., Nodarse, G., Gerhartz, A., 2013. Monitoreo de la anidación y estado de la conservación de las tortugas marinas en Cuba: Mayo 2010- Febrero 2012. Informe del Programa Cubano de Tortugas Marinas, del Programa de Conservación Marina de WWF en Cuba y del proyecto GEF-PNUD “Archipiélagos del Sur”. La Habana, 2012.

Azanza Ricardo, J., 2009. Estrategia reproductiva de la tortuga verde, *Chelonia mydas*, (Testudinata, Cheloniidae) y su impacto en la estructura genética de áreas de anidación del occidente del archipiélago cubano. Universidad de la Habana.

Azanza Ricardo, J., Cobian, D., Lopez, L.G., Pereda, C.G., Hernandez, N., 2013a. Results of the sixteenth season of the university project for the study and conservation of sea turtles in Guanahacabibes, Cuba. Technical report.

Azanza Ricardo, J., Ibarra Martín, M.E., González Sansón, G., Abreu-Grobois, F.A., Eckert, K.L., Espinosa López, G., Oyama Nakagawa, K., 2013b. Nesting ecology of *Chelonia mydas* (Testudines, Cheloniidae) on the Guanahacabibes Peninsula, Cuba. Int. J. Trop. Biol. Conserv. 61, 1935–1945.

Bachan, A., 2009. Position paper on sea turtle conservation in Trinidad and Tobago: 2009. Present. Int. Conf. Turt. Conserv. Ecotourism, Sustain. Community Dev. Univ. West Indies, July 28-29, 2009.

Bache, S.J., Frazier, J.G., 2006. International instruments and marine turtle conservation, in: Shanker, K., Choudhury, B.C. (Eds.), Marine Turtles of the Indian Subcontinent. Universities Press, pp. 324–353.

Bagda, E., Bardakci, F., Türkozan, O., 2012. Lower genetic structuring in mitochondrial DNA than nuclear DNA among the nesting colonies of green turtles (*Chelonia mydas*) in the Mediterranean. Biochem. Syst. Ecol. 43, 192–199.

Bagley, D.A., Kubis, S.A., Bresette, M.J., Ehrhart, L.M., 2008. Satellite tracking juvenile green turtles from Florida's east coast: the missing size classes found, in: Rees, A.F., Frick, M., Panagopoulou, A., Williams, K. (Eds.), Proceedings of the 27th Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-569, p. 37.

Baker, J.D., Littnan, C., Johnston, D., 2006. Potential effects of sea level rise on the terrestrial habitats of endangered and endemic megafauna in the Northwestern Hawaiian Islands. Endanger. Species Res. 2, 21–30.

Bal, G., Breheret, N., Vanleeuwe, H., 2007. An Update on Sea Turtle Conservation Activities in the Republic of Congo Conflicts and Social Dilemmas Associated with the Incidental Capture of Marine Turtles by Artisanal Fishers in Benin. Mar. Turt. Newsl. 9–10.

Balazs, G.H., 1975a. Marine turtles in the Phoenix Islands. Atoll Res. Bull. 184, 1–7.

Balazs, G.H., 1975b. Green turtle's uncertain future. Defenders 50, 521–523.

Balazs, G.H., 1976. Green turtle migrations in the Hawaiian archipelago. Biol. Conserv. 9, 125–140.

Balazs, G.H., 1980. Synopsis of biological data on the green turtle in the Hawaiian Islands. NOAA Tech. Memo. NMFS, NOAA-TM- NMFS-SWFC-7, and University of Hawaii Sea Grant Cooperative Report UNIHI-SEAGRANT CR-81-02.

Balazs, G.H., 1983. Sea turtles and their traditional usage in Tokelau. Atoll Res. Bull. 279, ii–30.

Balazs, G.H., 1985a. Impact of ocean debris on marine turtles: entanglement and ingestion, in: Shomura, R.S., Yoshida, H.O. (Eds.), Proceedings of the Workshop on the Fate and Impact of Marine Debris, 26-29 November 1984, Honolulu, Hawaii. NOAA Technical Memorandum NMFS/SWFC-54, pp. 387–429.

Balazs, G.H., 1985b. Status and ecology of marine turtles at Johnston Atoll. Atoll Res. Bull. 285, 1–46.

Balazs, G.H., 1993. Homeward bound: satellite tracking of Hawaiian green turtles from nesting beaches to foraging pastures. NOAA Tech. Memo. NMFS SEFSC.

Balazs, G.H., 1995. Status of Sea Turtles in the Central Pacific Ocean. Biol. Conserv. Sea Turtles 243–252.

Balazs, G.H., 1999. Historical summary of sea turtle observations at Rose Atoll, American Samoa, 1839-1993.

Balazs, G.H., 2000. Assessment of Hawaiian Green Turtles Utilizing Coastal Foraging Pastures at Palaau, Molokai, in: Bjorndal, K.A., Bolten, A.B. (Eds.), Proceedings of a Workshop on Assessing Abundance and Trends for In-Water Sea Turtle Populations. U.S. Dep. Commer. NOAA Tech. Mem. NMFS-SEFSC-44S, 83 p., pp. 42–44.

Balazs, G.H., Chaloupka, M.Y., 2004a. Thirty-year recovery trend in the once depleted Hawaiian green sea turtle stock. Biol. Conserv. 117, 491–498.

Balazs, G.H., Chaloupka, M.Y., 2004b. Spatial and temporal variability in somatic growth of green sea turtles (*Chelonia mydas*) resident in the Hawaiian Archipelago. Mar. Biol. 145, 1043–1059.

Balazs, G.H., Chaloupka, M.Y., 2006. Recovery trend over 32 years at the Hawaiian green turtle rookery of French Frigate Shoals. Atoll Res. Bull. 543, 147–158.

Balazs, G.H., Forsyth, R.G., Kam, A.K.H., 1987. Preliminary assessment of habitat utilization by Hawaiian Green Turtles in their resident foraging pastures.

Balazs, G.H., Kubis, S.A., 2007. Research plan for the Hawaiian green turtle, *Chelonia mydas*.

Balazs, G.H., Miya, R.K., Beavers, S.C., 1996. Procedures to attach a satellite transmitter to the carapace of an adult green turtle, *Chelonia mydas*, in: Keinath, J.A., Barnard, D.E., Musick, J.A., Bell, B.A. (Eds.), Fifteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-387, pp. 21–26.

Balazs, G.H., Murakawa, S.K.K., Wyneken, J., Schroeder, B.A., 1998. Differences in flipper size and esophagus morphology of green turtles from Hawaii and Florida, in: Epperly, S.P., Braun, J. (Eds.), Proceedings of the Seventeenth Annual Sea Turtle Symposium. U.S. Dep. Commer. NOAA Technical Memorandum NMFS-SEFSC-415, pp. 137–139.

Balazs, G.H., Rice, M.R., Hoffman, N., Murakawa, S.K.K., Parker, D.M., Shallenberger, R.J., 2005. Green Turtle foraging and resting habitats at Midway Atoll: significant findings over 25 years, 1975-2000, in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-528, pp. 102–104.

Balazs, G.H., Siu, P., Landret, J.P., 1995. Ecological aspects of green turtles nesting at Scilly Atoll in French Polynesia, in: Richardson, J.I., Richardson, T.H. (Eds.), Proceedings of the 12th Annual Workshop on Sea Turtle Biology and Conservation. U.S. Department of Commerce, NOAA Technical Memorandum NMFS, NMFS-SEFSC-361, pp. 7–10.

Baldwin, R., Hughes, G.R., Prince, R.I.T., 2003. Loggerhead turtles in the Indian Ocean. Loggerhead Sea Turtles. Smithson. Books, 218–232.

Banks, E., 1937. The breeding of the Edible Turtle (Chelone mydas). Sarawak Museum J. 4 (4), 523–532.

Baran, I., Durmus, S.H., Atatur, M.K., 1991. On *Chelonia mydas* (L.) (Reptilia:Chelonia) Population of Mersin-Kazanli Region. Turkish J. Zool. 15, 185–194.

Barata, P., González Carman, V., Dos Santos, A.S., Bondioli, A.C., Almeida, A.P., Santos, A.J.B., Da Silva, A.G.B., Giffoni, B., Domit, C., Baptistotte, C., Bellini, C., Batista, M., Bezerra, D., Monteiro, D., Albareda, D., Lima, H., Lima, E., Guebert-Bartholo, F., Sales, G., Lopez, G.G., Stahelin, G., Bruno, I., Castilhos, J., Thomé, J., Nunes, J., Becke, J., Wanderlinde, J., Rosa, L., Marcovaldi, M.Â., Melo, M., Mascarenhas, R., Estima, C., Naro-Maciel, E., 2011. Variação latitudinal na distribuição do tamanho de tartarugas-verdes (*Chelonia mydas*) ao longo da costa leste da América do Sul, in: Presentación En Las V Jornadas de Investigación Y Conservación de Tortugas Marinas Del Atlántico Sur Occidental. Florianópolis, Brasil, pp. 18–22.

Barbosa, C., Broderick, A.C., Catry, P., 1998. Marine Turtles in the Orango National Park (Bijagos Archipelago, Guinea-Bissau). Mar. Turt. Newsl. 81, 6–7.

Barnett, J., Adger, W.N., 2003. Climate dangers and atoll countries. Clim. Change 61, 321–337.

Barnett, L.K., Emms, C., Jallow, A., Cham, A.M., Mortimer, J.A., 2004. The distribution and conservation status of marine turtles in The Gambia, West Africa: a first assessment. Oryx 38, 203–208.

Barr, C., 2001. Current Status of Trade and Legal Protection for Sea Turtles in Indonesia. Mar. Turt. Newsl. 54, 4–7.

Barr, J.M., 2006. Community-based sea turtle monitoring and management at Helen Reef, Hatohobei State, Republic of Palau. Oregon State University.

Bartholomew, D.J., 1965. A comparison of some bayesian and frequenist inferences. Biometirika 52, 19–35.

Bartram, P.K., Kaneko, J.J., 2004. Catch to bycatch ratios: Comparing Hawaii's longline fishery with others. SOEST Publication 04-05, JIMAR Contribution 04-352, 40 pp.

Basintal, P., 2002. Sea turtle conservation at the Sabah's Turtle lslands Park, Malaysia, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. February 5–8, 2002. Honolulu, HI: Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 151–161.

Bass, A.L., Epperly, S.P., Braun-McNeill, J., 2004. Multi-year analysis of stock composition of a loggerhead turtle (*Caretta caretta*) foraging habitat using maximum likelihood and Bayesian methods. Conserv. Genet. 5.

Bass, A.L., Epperly, S.P., Braun-McNeill, J., 2006. Green turtle (*Chelonia mydas*) foraging and nesting aggregations in the Caribbean and Atlantic: impact of currents and behavior on dispersal. J. Hered. 97, 346.

Bass, A.L., Lagueux, C.J., Bowen, B.W., 1998. Origin of green turtles, *Chelonia mydas*, at "Sleeping Rocks" off the northeast coast of Nicaragua. Copeia 1998, 1064–1069.

Bass, A.L., Witzell, W.N., 2000. Demographic Composition of Immature Green Turtles (*Chelonia mydas*) from the East Central Florida Coast: Evidence from mtDNA Markers. Herpetologica 56, 357–367.

Basson, P.W., Burchard Jr., J.E., Hardy, J.T., Price, A.R.G., 1977. Biotopes of the western Arabian Gulf: marine life and environments of Saudi Arabia. Aramco Department of Loss Prevention and Environmental Affaris, Great Britan.

Batibasaga, A., Waqainabete, S., Qauqau, A., 2006. Notes on Fijian sea turtles: Estimates on population status. Information provided for Sea Turtle Working Group Meeting-Nadave/CATD 31st May- 1st June.

Bell, C.D., Austin, T., 2003. The continued assessment of the reproductive status of the marine turtle rookery in the Cayman Islands, in: Seminoff, J.A. (Ed.), Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503. 308 pages, pp. 87–88.

Bell, C.D., Blumenthal, J.M., Broderick, A.C., Godley, B.J., 2010. Investigating potential for depensation in marine turtles: how low can you go? Conserv. Biol. 24, 226–235.

Bell, C.D., Parsons, J., Austin, T.J., Broderick, A.C., Ebanks-Petrie, G., Godley, B.J., 2005. Some of them came home: the Cayman Turtle Farm headstarting project for the green turtle *Chelonia mydas*. Oryx 39, 137–148.

Bell, C.D., Solomon, J.L., Blumenthal, J.M., Austin, T.J., Ebanks-Petrie, G., Broderick, A.C., Godley, B.J., 2007. Monitoring and conservation of critically reduced marine turtle nesting populations: lessons from the Cayman Islands. Anim. Conserv. 10, 39–47.

Bell, L.A.J., Fa'anunu, U., Koloa, T., 1994. Fisheries resources profiles: Kingdom of Tonga. FFA Report 94/5.

Bell, L.A.J., Matato, L., Fa'anunu, U., 2009. Project report marine turtle monitoring programme in Tonga. Marine Conservation Action Fund, 15 pages plus appendices.

Bellini, C., Marcovaldi, M.Â., Sanches, T.M., Grossman, A., Sales, G., 1996. Atol das Rocas biological reserve: second largest Chelonia rookery in Brazil. Mar. Turt. Newsl. 72, 1–2.

Bellini, C., Santos, A.J.B., Grossman, A., Marcovaldi, M.Â., Barata, P.C.R., 2013. Green turtle (*Chelonia mydas*) nesting on Atol das Rocas, north-eastern Brazil, 1990-2008. J. Mar. Biol. Assoc. United Kingdom 93, 1117–1132.

Bellmund, S.A., Musick, J.A., Klinger, R.C., Byles, R.A., Keinath, J.A., Barnard, D.E., 1987. Ecology of sea turtles in Virginia. Virginia Institue Mar. Sci. Coll. William Mary, Gloucester Point, Virginia.

Berdach, J.T., Mandeakali, L., 2004. Regional: mainstreaming environmental considerations in economic and development planning processes in selected Pacific developing member countries. ADB Technical Assistance Consultant's Report, Project Number 38031, December 2004.

Berger, K., 1991. The impacts of South China Sea oil spills on the environment of Palawan. Philipp. Sci. 28, 27–54.

Bhaskar, S., 1979. Sea turtle survey in the Andaman and Nicobars. Hamadryad 4, 2–26.

Bhaskar, S., 1984. The distribution and status of sea turtles in India, in: Silas, E.G. (Ed.), Proceedings of the Workshop on Sea Turtle Conservation. CMFRI Special Publication 18, pp. 21–35.

Bhaskar, S., 1993. The status and ecology of sea turtles in the Andaman and Nicobar Islands. Centre for Herpetology Publication No. ST 1/93: 1-37.

Bhupathy, S., Saravanan, S., 2006. Status of marine turtles in the Gulf of Mannar, India. Chelonian Conserv. Biol. 5, 139–141.

Binckley, C.A., Spotila, J.R., Wilson, K.S., Paladino, F. V., 1998. Sex determination and sex ratios of Pacific leatherback turtles, *Dermochelys coriacea*. Copeia 1998, 291–300.

BIOMARCC-SINAC-GIZ, 2103. Estudios cientificos de habitat marino costero y situacion socioeconomic del Pacific Norte de Costa Rica. San Jose-Costa Rica.

Björk, M., Short, F., Mcleod, E., Beer, S., 2008. Managing seagrasses for resilience to climate change. IUCN, Gland, Switzerland.

Bjorkland, R., Thomas, C., Hutchinson, J., Crowder, L., 2008. Preliminary survey of incidental capture of sea turtles in Jamaican fisheries, in: Rees, A.F., Frick, M., Panagopoulou, A., Williams, K. (Eds.), Proceedings of the 27th Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-569. 262 pages, p. 167.

Bjorndal, K.A., 1980. Nutrition and grazing behaviour of the green turtle, *Chelonia mydas*. Mar. Biol. 56, 147–154.

Bjorndal, K.A., 1982. The consequences of Herbivory for the Life History Pattern of the Caribbean Green Turtle, *Chelonia mydas*, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, pp. 111–116.

Bjorndal, K.A., 1997. Foraging ecology and nutrition of sea turtles, in: Lutz, P.L., Musick, J.A. (Eds.), The Biology of Sea Turtles. CRC Press Boca Raton, Florida, pp. 199–232.

Bjorndal, K.A., 2008. Biology of immature green turtles on neritic foraging grounds: insights from a 30-year study in the southern Bahamas, in: Kalb, H.J., Rohde, A., Gayheart, K., Shanker, K. (Eds.), Proceedings of the Twenty-Fifth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-582. 204 pages, p. 22.

Bjorndal, K.A., Bolten, A.B., 1996. Developmental migrations of juvenile green turtles in the Bahamas, in: Keinath, J.A., Barnard, D.E., Musick, J.A., Bell, B.A. (Eds.), Proceedings of the Fifteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-387. 355 pages, pp. 38–39.

Bjorndal, K.A., Bolten, A.B., 2008. Annual variation in source contributions to a mixed stock: implications for quantifying connectivity. Mol. Ecol. 17, 2185–2193.

Bjorndal, K.A., Bolten, A.B., 2010. Policy changes protect sea turtles in The Bahamas: Long-term efforts rewarded. State of the World's Turtles 5, 17.

Bjorndal, K.A., Bolten, A.B., Chaloupka, M.Y., 2000. Green Turtle Somatic Growth Model: Evidence Fordensity Dependence. Ecol. Appl. 10, 269.

Bjorndal, K.A., Bolten, A.B., Chaloupka, M.Y., 2003. Survival probability estimates for immature green turtles *Chelonia mydas* in the Bahamas. Mar. Ecol. Prog. Ser. 252, 273–281.

Bjorndal, K.A., Bolten, A.B., Lagueux, C.J., 1994. Ingestion of marine debris by juvenile sea turtles in coastal Florida habitats. Mar. Pollut. Bull. 28, 154–158.

Bjorndal, K.A., Bolten, A.B., Moreira, L., Bellini, C., Marcovaldi, M.Â., 2006. Population Structure and Diversity of Brazilian Green Turtle Rookeries Based on Mitochondrial DNA Sequences. Chelonian Conserv. Biol. 5, 262.

Bjorndal, K.A., Wetherall, J.A., Bolten, A.B., Mortimer, J.A., 1999. Twenty-six years of green turtle nesting at Tortuguero, Costa Rica: An encouraging trend. Conserv. Biol. 13, 126–134.

Blanco, G.S., 2010. Movements and Behavior of the East Pacific Green Turtle (*Chelonia mydas*) from Costa Rica. PhD thesis, Drexel University.

Blanco, G.S., Morreale, S.J., Bailey, H., Seminoff, J.A., Paladino, F. V., Spotila, J.R., 2012. Post-nesting movements and feeding grounds of a resident East Pacific green turtle *Chelonia mydas* population from Costa Rica. Endanger. Species Res. 18, 233–245.

Blanco, G.S., Santidrián, P., 2011. La Tortuga marina negra de Guanacase amenazada de extincion por practicas humanas, in: Tortugas Marinas En Costa Rica: "enemigos", Proteccion, Y Aprocamiento Sostenible. Special issue of Ambientales. Universidad National Costa Rica, San Jose., pp. 19–25.

Blanco, R., Escobar, E., Forneiro, Y., Moncada, F.G., Medina, Y., 2009. Annual report of research and conservation of marine turtles. Havana, Cuba: Fisheries Research Center for the Ministry of Fisheries. Final report of Sea Turtles in Guanahacabibes, Cuba. Technical report.

Blanco, W., Cardona, R., 1983. Las pesquerias de tortugas marinas y algunos aspectos biologicos pesqueros en el Archipielago Cubano. Resumenes IV Foro Cient. Cent. Investig. Pesq. CIP-MIP.

Blumenthal, J.M., Solomon, J.L., Bell, C.D., Austin, T.J., Ebanks-Petrie, G., Coyne, M.S., Broderick, A.C., Godley, B.J., 2006. Satellite tracking highlights the need for international cooperation in marine turtle management. Endanger. Species Res. 2, 51–61.

Bolker, B., Okayama, T., Bjorndal, K.A., Bolten, A.B., 2007. Incorporating multiple mixed stocks in mixed stock analysis: "many-to-many" analyses. Mol. Ecol. 16, 685–695.

Bolten, A.B., 2003. Variation in sea turtle life history patterns: Neritic vs. oceanic developmental stages, in: Lutz, P.L., Musick, J.A., Wyneken, J. (Eds.), The Biology of Sea Turtles, Volumen II. CRC Press Boca Raton, FL, p. 455.

Bool, N., Whiting, A.U., Gourlay, T., Mau, R., 2009. Ningaloo Turtle Program Annual Report 2008-2009, Program. Ningaloo Turtle Program, Exmouth, Western Australia.

Bornatowski, H., Heithaus, M.R., Batista, C.M.P., Mascarenhas, R., 2012. Shark scavenging and predation on sea turtles in northeastern Brazil. Amphibia-Reptilia 33, 495–502.

Borrowman, K.M., 2008. Prevalence and severity of fibropapilomatoasis in juvenile green turtles (*Chelonia mydas*) in three habitats on Florida's east coast. University of Central Florida.

Bouchard, S., Moran, K., Tiwari, K., Wood, D., Bolten, A.B., Eliazar, P., Bjorndal, K.A., 1998. Effects of exposed pilings on sea turtle nesting activity at Melbourne Beach, Florida. J. Coast. Res. 14, 1343–1347.

Bourjea, J., 2012. Sea turtles: a review of the status, distribution, and interaction with fisheries in the southwest Indian Ocean, in: van der Elst, R. (Ed.), Mainstreaming Biodiversity: A SWIOFP Retrospective Analysis. South West Indian Ocean Fisheries Project (SWIOFP), pp. 35–63.

Bourjea, J., Ciccione, S., Ratsimbazafy, R., 2006. Marine Turtles Surveys in Nosy Iranja Kely, North-Western. West. Indian Ocean J. Mar. Sci. 5, 209–212.

Bourjea, J., Dalleau, M., Ciccione, S., 2013. Seasonal variability of migrating corridors and foraging areas of adult green turtles revealed by satellite tracking at the regional scale, in: Proceedings of the 33rd Annual Symposium on Sea Turtle Biology and Conservation. Baltimore, Maryland.

Bourjea, J., Frappier, J., Quillard, M., Ciccione, S., Roos, D., Hughes, G.R., Grizel, H., 2007a. Mayotte Island: another important green turtle nesting site in the southwest Indian Ocean. Endanger. Species Res. 3, 273–282.

Bourjea, J., Lapègue, S., Gagnevin, L., Broderick, D., Mortimer, J.A., Ciccione, S., Roos, D., Taquet, C., Grisel, H., 2007b. Phylogeography of the green turtle, *Chelonia mydas*, in the Southwest Indian Ocean. Mol. Ecol. 16, 175–186.

Bourjea, J., Nel, R., Jiddawi, N.S., Koonjul, M.S., Bianchi, G., 2008. Sea turtle bycatch in the West Indian Ocean: review, recommendations and research priorities. West. Indian Ocean J. Mar. Sci. 7, 137–150.

Bovery, C.M., Wyneken, J., 2013. Sea turtles in Florida's Atlantic waters. Mar. Fish. Rev. 75, 1–12.

Bowen, B.W., 2003. What Is A Loggerhead Turtle? The Genetic Perspective, in: Bolten, A.B., Witherington, B.E. (Eds.), Loggerhead Sea Turtles. Smithsonian Books, pp. 7–27.

Bowen, B.W., Karl, S.A., 2007. Population genetics and phylogeography of sea turtles. Mol. Ecol. 16, 4886–4907.

Bowen, B.W., Meylan, A.B., Ross, J.P., Limpus, C.J., Balazs, G.H., Avise, J.C., 1992. Global Population Structure and Natural History of the Green Turtle (*Chelonia mydas*) in Terms of Matriarchal Phylogeny. Evolution (N. Y). 46, 865–881.

Boyle, M.C., Limpus, C.J., 2008. The stomach contents of post-hatchling green and loggerhead sea turtles in the southwest Pacific: an insight into habitat association. Mar. Biol. 155, 233–241.

Bradai, M.N., Jribi, I., 2010. Tunisia, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Priorities. IUCN, Gland, Switzerland, pp. 245–256.

Bradshaw, P., Bradshaw, N., 2012. Turtles and their habitats on Mauke, Southern Cook Islands. Pacific Islands Conservation Initiative (PICI), Cook Islands Turtle Project: Mauke Report 2012.

Brainard, R.E., Birkeland, C., Eakin, C.M., McElhany, P., Miller, M.W., Patterson, M., Piniak, G.A., 2011. Status review report of 82 candidate coral species petitioned under the U.S. Endangered Species Act. U.S. Dep. Commer., NOAA Tech. Memo., NOAA-TM-NMFS-PIFSC-27.

Brattstrom, B.H., 1982. Breeding of the green sea turtle, *Chelonia mydas*, on the Islas Revillagigedo, Mexico. Herpetol. Rev. 13, 71.

Bräutigam, A., Eckert, K.L., 2006. Turning the tide: exploitation, trade and management of marine turtles in the Lesser Antilles, Central America, Colombia and Venezuela, in: TRAFFIC International and the Secretariat of the Convention on International Trade in Endangered Species.

Bresette, M., Gorham, J., Peery, B., 1998. Site fidelity and size frequencies of juvenile green turtles (*Chelonia mydas*) utilizing near shore reefs in St. Lucie County, Florida. Mar. Turt. Newsl. 82, 5–7.

Bresette, M.J., Gorham, J.C., Peery, B.D., 2002. Initial assessment of sea turtles in the southern Indian River Lagoon System, Ft. Pierce Florida., in: Mosier, A., Foley, A., Brost, B. (Eds.), Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. Memo. NMFS-SEFSC- 477, pp. 271–273.

Bresette, M.J., Herren, R.M., Singewald, D.A., 2005. Comparison of fibropapilloma rates of green turtles (*Chelonia mydas*) from two different sites in St. Lucie County, Florida, in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-528. 3368 pages, pp. 125–126.

Bresette, M.J., Witherington, B.E., Herren, R.M., Bagley, D.A., Gorham, J.C., Traxler, S.L., Crady, C.K., Hardy, R., 2010. Size-class partitioning and herding in a foraging group of green turtles *Chelonia mydas*. Endanger. Species Res. 9, 105–116.

Briggs, J.C., 1974. Marine Zoogeography. McGraw Hill, New York.

Brikke, S., 2009. Local perceptions of sea turtles on Bora Bora and Maupiti Islands, French Polynesia. SPC Tracitional Marine Resource Managment and Knowledge Information Bulletin #26.

Broderick, A.C., Coyne, M.S., Fuller, W.J., Glen, F., Godley, B.J., 2007. Fidelity and over-wintering of sea turtles. Proc. Biol. Sci. 274, 1533–8.

Broderick, A.C., Frauenstein, R., Glen, F., Hays, G.C., Jackson, A.L., Pelembe, T., Ruxton, G.D., Godley, B.J., 2006. Are green turtles globally endangered? Glob. Ecol. Biogeogr. 15, 21–26.

Broderick, A.C., Glen, F., Godley, B.J., Hays, G.C., 2002a. Estimating the number of green and loggerhead turtles nesting annually in the Mediterranean. Oryx 36, 227–235.

Broderick, A.C., Glen, F., Godley, B.J., Hays, G.C., 2002b. A management plan for the marine turtles of Ascension Island. Marine Turtle Research Group, University of Wales Swansea, Swansea, U.K.

Broderick, A.C., Glen, F., Godley, B.J., Hays, G.C., 2003. Variation in reproductive output of marine turtles. J. Exp. Mar. Bio. Ecol. 288, 95–109.

Broderick, A.C., Godley, B.J., 1994. Marine Turtles in Northern Cyprus: Results From Glasgow University Turtle Conservation Expeditions, 1992-1993. Mar. Turt. Newsl. 67, 8–11.

Broderick, A.C., Godley, B.J., Hays, G.C., 2001. Metabolic heating and the prediction of sex ratios for green turtles (*Chelonia mydas*). Physiol. Biochem. Zool. 74, 161–170 ST – Metabolic heating and the prediction.

Broderick, A.C., Godley, B.J., Reece, S., Downie, J.R.R., 2000. Incubation periods and sex ratios of green turtles: highly female biased hatchling production in the eastern Mediterranean. Mar. Ecol. Prog. Ser. 202, 273–281.

Broderick, A.C., Hancock, E.G., 1997. Marine Turtle Eggs. Herpetol. Rev. 28, 190–191.

Broderick, D., 1998. Subsistence harvesting of marine turtles in the Solomon Islands, in: Epperly, S.P., Braun, J. (Eds.), Proceedings of the Seventeenth Annual Sea Turtle Symposium. NOAA Technical Memorandum NMFS-SEFSC-415, pp. 15–18.

Brongersma, L.D., 1982. Marine turtles of the eastern Atlantic Ocean, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, pp. 407–416.

Brooke, M. de L., 1995. Seasonality and numbers of green turtles *Chelonia mydas* nesting on the Pitcairn Islands. Biol. J. Linn. Soc. 56, 325–327.

Brown, L., MacDonald, D.W., 1995. Predation on green turtle *Chelonia mydas* nests by wild canids at Akyatan Beach, Turkey. Biol. Conserv. 71, 55–60.

Buden, D.W., 1999. Reptiles, birds, and mammals of Oroluk Atoll, Eastern Caroline Islands. Micronesica 31, 289–300.

Buden, D.W., 2008. The reptiles of Nauru. Pacific Sci. 62, 499–507.

Buden, D.W., 2013. Reptiles of Sorol Atoll, Yap, Federated States of Micronesia. Pacific Sci. 67, 119–128.

Buden, D.W., Edward, A., 2001. Abundance and utilization of sea turtles on Pohnpei, Federated States of Micronesia: Islanders' perceptions. Micronesica 34, 47–54.

Bugoni, L., Krause, L., Petry, M. V., 2001. Marine debris and human impacts on sea turtles in southern Brazil. Mar. Pollut. Bull. 42, 1330–1334.

Bureau of Marine Resources, 2005. Palau marine turtle conservation and management project, September 2004 to September 2005. Bureau of Marine Resources, Republic of Palau.

Burton, A., 2012. Glad tidings for green sea turtles. Front. Ecol. Environ. 10, 63.

Buss, P.E., Bengis, R.G., 2012. Cyanobacterial biointoxication in free-ranging wildlife, in: Miller, R.E., Fowler, M. (Eds.), Fowler's Zoo and Wild Animal Medicine. Elsevier Inc., Saint Louis, pp. 108–114.

Bustard, R., 1974. Barrier reef sea turtles populations. Proc. Second Int. coral reef Symp. 227–234.

Camiñas, J.A., 2004. Sea turtles of the Mediterranean Sea: population dynamics, sources of mortality and relative importance of fisheries impacts, in: Papers Presented at the Expert Consultation on Interactions between Sea Turtles and Fisheries within an Ecosystem Context. FAO Fisheries Report. No. 738, Supplement, Rome, Italy, pp. 27–84.

Campbell, C.L., Lagueux, C.J., 2005. Survival probability estimates for large juvenile and adult green turtles (*Chelonia mydas*) exposed to an artisanal marine turtle fishery in the Western Caribbean. Herpetologica 61, 91–103.

Canbolat, A.F., Atatunc, K., Candan, O., Barcak, D., 2009. A new green turtle (*Chelonia mydas*) nesting site in the Mediterranean: Sugozu beaches, Adana (Turkey), in: Demetropoulos, A., Turkozan, O. (Eds.), Proceedings of the Second Mediterranean Conference on Marine Turtles. Barcelona Convention – Bern Convention – Bonn Convention (CMS), Kemer, Turkey, p. 65.

Carballo, J.L., Olabarria, C., Garza Osuna, T., 2002. Analysis of four macroalgal assemblages along the Pacific Mexican coast during and after the 1997-98 El Nino. Ecosystems 5, 749–760.

Carr, A.F., 1952. Genus Lepidochelys - The ridleys, Handbook of turtles.

Carr, A.F., 1961. Pacific turtle problem. Nat. Hist. 70, 64–71.

Carr, A.F., 1987. Impact of nondegradable marine debris on the ecology and survival outlook of sea turtles. Mar. Pollut. Bull. 18, 352–356.

Carr, A.F., Carr, M.H., Meylan, A.B., 1978. The ecology and migrations of sea turtles, 7. The West Caribbean green turtle colony. Bull. Am. Museum Nat. Hist. 162, 1–46.

Carr, A.F., Meylan, A.B., Mortimer, J.A., Bjorndal, K.A., Carruthers, T.J.B., Carr, T., 1982. Surveys of sea turtle populations and habitats in the western Atlantic, US Department of Commerce NOAA Tech Memo. NMFS-SEFC-91. 91pp.

Carr, A.F., Ogren, L.H., 1960. The ecology and migrations of sea turtles, 4: the green turtle in the Caribbean Sea. Bull. Am. Museum Nat. Hist. 121, 1–48.

Carr, T., Carr, N., 1991. Surveys of the sea-turtles of Angola. Biol. Conserv. 58, 19–29.

Carrión-Cortez, J.A., Zárate, P., Seminoff, J.A., 2010. Feeding ecology of the green sea turtle (*Chelonia mydas*) in the Galapagos Islands. J. Mar. Biol. Assoc. United Kingdom 90, 1005–1013.

Casale, P., 2008. Incidental catch of marine turtles in the Mediterranean Sea: captures, mortality, priorities. Rome, Italy.

Casale, P., 2010. Italy, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Priorities. IUCN, Gland, Switzerland, pp. 135–148.

Casale, P., 2011. Sea turtle by-catch in the Mediterranean. Fish Fish. 12, 299–316.

Casale, P., Cattarino, L., Freggi, D., Rocco, M., Argano, R., 2007. Incidental catch of marine turtles by Italian trawlers and longliners in the central Mediterranean. Aquat. Conserv. Mar. Freshw. Ecosyst. 17, 686–701.

Casale, P., Gerosa, G., Yerli, S. V., 2000. Female-biased primary sex ratio of the Green Turtle, *Chelonia mydas*, estimated through sand temperatures at Akyatan, Turkey. Zool. Middle East 20, 33–42 ST – Female–biased primary sex ratio of the.

Casale, P., Laurent, L., De Metrio, G., 2004. Incidental capture of marine turtles by the Italian trawl fishery in the north Adriatic Sea. Biol. Conserv. 119, 287–295.

Casale, P., Margaritoulis, D., 2010. Sea turtles in the Mediterranean: Distribution, threats and conservation policies. Gland, Switzerland.

Casale, P., Margaritoulis, D., Aksissou, M., Aureggi, M., Benhardouze, W., Bradai, M.N., Broderick, A.C., Carreras, C., Demetropoulos, A., Fuller, W.J., Glen, F., Godfrey, M.H., Hadjichristophorou, M., Hamza, A., Haxhiu, I., Jony, M., Jribi, I., Kaska, Y., Khalil, M., Kusetoğullari, H., Lazar, B., Levy, Y., Mifsud, C., Nada, M., Oliver, G., Panagopoulou, A., Rees, A.F., Saad, A., Tiwari, M., Tomás, J., Türkozan, O., Ziza, V., 2010. Overview, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Priorities. International Union for Conservation of Nature and Natural Resources, Gland, Switzerland, pp. 1–14.

Castroviejo, J., Juste, J.B., del Val, J.P., Castelo, R., Gil, R., 1994. Diversity and status of sea turtles species in the Gulf of Guniea islands. Biodivers. Conserv. 826–836.

Catry, P., Barbosa, C., Indjai, B., 2010. Marine turtles of Guinea-Bissau. Status, biology and conservation. Instituto da Biodiversidade e das Áreas Protegidas, Bissau.

Catry, P., Barbosa, C., Indjai, B., Almeida, A., Godley, B.J., Vié, J.-C., 2002. First census of the green turtle at Poilão, Bijagós Archipelago, Guinea-Bissau: the most important nesting colony on the Atlantic coast of Africa. Oryx 36, 400–403.

Catry, P., Barbosa, C., Paris, B., Indjai, B., Almeida, A., Limoges, B., Silva, C., Pereira, H., 2009. Status, Ecology, and Conservation of Sea Turtles in Guinea-Bissau. Chelonian Conserv. Biol. 8, 150–160.

Caverhill, B., Noto, A., Chan, E.H., Chen, P.N., 2012. Sea turtle reproduction [WWW Document]. Turt. Conserv. Soc. Malaysia. URL http://www.turtleconservationsociety.org.my/posters (accessed 10.16.12).

Cayman Islands Government, 1978. Marine Conservation (Turtle Protection) Regulations, 1978.

Cayman Islands Government, 2008. The Marine Conservation (Turtle Protection) (Amendment) Regulations (2008 revision).

Chacón, D., 2002. Assessment about the trade of sea turtles and their products in the Central American isthmus. Red Regional para la Conservación de last Tortugas Marinas en Centroamérica, San José, Costa Rica.

Chacón, D., Arauz, R., 2001. Diagnostico regional y planificacion estrategica para la conservacion de las tortugas marinas en Centroamerica. Red Reg. para la Conserv. las tortugas Mar. Centroam. 134.

Chaloupka, M.Y., 2002. Stochastic simulation modelling of southern Great Barrier Reef green turtle population dynamics. Ecol. Modell. 148, 79–109.

Chaloupka, M.Y., Balazs, G.H., 2005. Modelling the effect of fibropapilloma disease on the somatic growth dynamics of Hawaiian green sea turtles. Mar. Biol. 147, 1251–1260.

Chaloupka, M.Y., Balazs, G.H., Work, T.M., 2009. Rise and fall over 26 years of a marine epizootic in Hawaiian green sea turtles. J. Wildl. Dis. 45, 1138–1142.

Chaloupka, M.Y., Bjorndal, K.A., Balazs, G.H., Bolten, A.B., Ehrhart, L.M., Limpus, C.J., Suganuma, H., Troëng, S., Yamaguchi, M., 2008. Encouraging outlook for recovery of a once severely exploited marine megaherbivore. Glob. Ecol. Biogeogr. 17, 297–304.

Chaloupka, M.Y., Limpus, C.J., 2001. Trends in the abundance of sea turtles resident in southern Great Barrier Reef waters. Biol. Conserv. 102, 235–249.

Chaloupka, M.Y., Limpus, C.J., 2005. Estimates of sex- and age-class-specific survival probabilities for a southern Great Barrier Reef green sea turtle population. Mar. Biol. 146, 1251–1261.

Chaloupka, M.Y., Limpus, C.J., Miller, J.D., 2004. Green turtle somatic growth dynamics in a spatially disjunct Great Barrier Reef metapopulation. Coral Reefs 23, 325–335.

Chaloupka, M.Y., Musick, J.A., 1997. Age, growth, and population dynamics, in: Lutz, P.L., Musick, J.A. (Eds.), The Biology of Sea Turtles. CRC Press, pp. 233–273.

Chan, E.H., 2004. Turtles in Trouble. Siri Syarahan Inaug. KUSTEM 7, 1–27.

Chan, E.H., 2006. Marine turtles in Malaysia: on the verge of extinction? Aquat. Ecosyst. Health Manag. 9, 175–184.

Chan, E.H., 2010. A 16-year record of green and hawksbill turtle nesting activity at Chagar Hutang Turtle Sanctuary, Redang Island, Malaysia. Indian Ocean Turt. Newsl. 1–5.

Chan, E.H., Liew, H.-C., 1995. Incubation temperatures and sex-ratios in the Malaysian leatherback turtle *Dermochelys coriacea*. Biol. Conserv. 74, 169–174.

Chan, E.H., Liew, H.-C., Mazlan, A.G., 1988. The incidental capture of sea turtles in fishing gear in Terengganu, Malaysia. Biol. Conserv. 43, 1–7.

Chan, S.K.F., Cheng, I.-J., Zhou, T., Wang, H.J., Gu, H.X., Song, X.J., 2007. A comprehensive overview of the population and conservation status of sea turtles in China. Chelonian Conserv. Biol. 6, 185–198.

Chandi, M., Deol, R., Shetty, R.S., 2012. Socioeconomic Monitoring for Coastal Managers of South Asia: Field Trials and Baseline Surveys, Havelock and Neil Islands, Rani Jhansi Marine National Park, Andaman Islands, ANET, India.

Chantrapornsyl, S., 1993. Status of marine turtles in Thailand, in: Proceedings of the First ASEAN Symposium-Workshop on Marine Turtle Conservation, Manila, Philippines. pp. 123–128.

Charilaou, P., Perdiou, A., 2012. Marine-turtle survey at Akrotiri-Episkopi, 2010-11, version 1.3. SBAA Environment Department, Episkopi.

Charlier, P., Ijspol, M., Babb-Echteld, Y., Booth, A., 2000. Shrimp and finfish fisheries in Suriname, in: Report of the Third Workskop on the Assessment of Shrimp and Groundfish Fisheries on the Brazil-Guinas Shelf. Belem, Brazil, 24 May - 10 June 1999. FAO Fisheries Report. No. 628., FAO.

Charuchinda, M., Monanunsap, S., 1998. Monitoring survey on sea turtle nesting in the Inner Gulf of Thailand, 1994-1996. Thai Mar. Fish. Res. Bull. 6, 17–25.

Charuchinda, M., Monanunsap, S., Chantrapornsyl, S., 2002. Status of sea turtle conservation in Thailand, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. February 5–8, 2002. Honolulu, HI: Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 179–184.

Charuchinda, M., Sakamoto, W., Arai, N., Monanunsap, N., 2003. Migration pattern of post-nesting green turtles in the Gulf of Thailand, in: Arai, N. (Ed.), Proceedings of the 3rd Workship on SEASTAR2000. pp. 53–57.

Chatto, R., Baker, B., 2008. The distribution and status of marine turtle nesting in the Northern Territory.

Chatto, R., Guinea, M.L., Conway, S., 1995. Sea turtles killed by flotsam in northern Australia. Mar. Turt. Newsl. 69, 17–18.

Chen, C.-L., Wang, C.-C., Cheng, I.-J., 2010. Effects of biotic and abiotic factors on the oxygen content of green sea turtle nests during embryogenesis. J. Comp. Physiol. B. 180, 1045–55.

Chen, T.H., Cheng, I.-J., 1995. Breeding biology of the green turtle, *Chelonia mydas*, (Reptilia: Cheloniidae) on Wan-An Island, Peng-Hu Archipelago, Taiwan. Mar. Biol. 124, 9–15.

Cheng, I.-J., 1995. Tourism and the green turtle in conflict on Wan-An Island, Taiwan. Mar. Turt. Newsl. 68, 4–6.

Cheng, I.-J., 1996. Sea turtles at Taipin Tao, South China Sea. Mar. Turt. Newsl. 6–8.

Cheng, I.-J., 2000. Post-nesting migrations of green turtles (*Chelonia mydas*) at Wan-An Island, Penghu Archipelago, Taiwan. Mar. Biol. 137, 747–754.

Cheng, I.-J., Dutton, P.H., Chen, C.-L., Chen, H.-C., Chen, Y.-H., Shea, J.-W., 2008. Comparison of the genetics and nesting ecology of two green turtle rookeries. J. Zool. 276, 375–384.

Cheng, I.-J., Huang, C.T., Hung, P.Y., Ke, B.Z., Kuo, C.W., Fong, C.L., 2009. Ten years of monitoring the nesting ecology of the green turtle, *Chelonia mydas*, on Lanyu (Orchid Island), Taiwan. Zool. Stud. 48, 83–94.

Ciccione, S., Bourjea, J., 2006. Nesting of green turtles in Saint Leu, Réunion Island. Mar. Turt. Newsl. 112, 1–3.

Ciccione, S., Taquet, M., Roos, D., Barde, J., 2002. Assistance a la DAF de Mayotte pour la mise en place d'une etude sur les tortues marines, Rapport final CEDTM-IFREMER 35 p.

Clapp, R.B., Wirtz, W.O., 1975. The natural history of Lisianski Island, Northwestern Hawaiian Islands. Atoll Res. Bull. 186, 1–196.

Clark, R.R., 1992. Beach conditions in Florida: a statewide inventory and identification of beach erosion problem areas in Florida, in: Beaches and Shores Technical and Design Memorandum 89-1, 4th Edition. Florida Deparment of Natural Resources, Division of Beaches and Shores, Tallahassee, Florida, p. 208.

Clarke, M., Campbell, A.C., Hameid, W.S., Ghoneim, S., 2000. Preliminary report on the status of marine turtle nesting populations on the Mediterranean coast of Egypt. Biol. Conserv. 94, 363–371.

Cliffton, K., Cornejo, D.O., Felger, R.S., 1982. Sea turtles of the pacific coast of Mexico, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington, D.C., pp. 199–209.

Collazo, J.A., R., B.J., Tallevast, T.L., 1992. Abundance and growth patterns of *Chelonia mydas* in Culebra, Puerto Rico. J. Herpetol. 26, 293–300.

Commonwealth of the Northern Mariana Islands Coastal Resources Management Office, 2011. Section 309 assessment and strategy report 2011-2015.

Commonwealth of the Northern Mariana Islands Division of Fish and Wildlife, 2009. Population dynamics of sea turtles at the Northern Marianas. Annual Report to NMFS PIRO: October 1, 2008 to September 30, 2009. Grant Number NA08NMF4540613. 222 pages.

Conant, T.A., Dutton, P.H., Eguchi, T., Epperly, S.P., Fahy, C.C., Godfrey, M.H., MacPherson, S.L., Possardt, E.E., Schroeder, B.A., Seminoff, J.A., Snover, M.L., Upite, C.M., Witherington, B.E., 2009. Loggerhead sea turtle (*Caretta caretta*) 2009 status review under the US Endangered Species Act, Report of the Biological Review Team to the National Marine Fisheries Service.

Conservation International Pacific Islands Program, 2013. Biodiversity Conservation Lessons Learned Technical Series 11: Double programme de recherche sur les tortues marines de l'Archipel de la Société, Polynésie française [Dual Research Program on Sea Turtles of the Society Archipelago – French Polynesia].

Costa, A., Motta, H., Pereira, M.A.M., Videira, E.J.S., Louro, C.M.M., Joao, J., 2007. Marine turtles in Mozambique: towards an effective conservation and management program. Mar. Turt. Newsl. 117, 1–3.

Cox, J.H., Percival, H.F., Colwell, S.V., 1994. Impact of vehicular traffic on beach habitat and wildlife at Cape San Blas, Florida.

Coyne, M.S., 1994. Nesting ecology of subadult green sea turtles in south Texas waters. Master of Science, Texas A&M.

Craig, P., 1993. Sea turtles: A vanishing resource in American Samoa. Mar. Turt. Newsl. 61, 13–15.

Craig, P., Balazs, G.H., 1995. Marine Turtle Travels from American Samoa to French Polynesia. Mar. Turt. Newsl. 70, 5–6.

Craig, P., Parker, D.M., Brainard, R.E., Rice, M., Balazs, G.H., 2004. Migrations of green turtles in the central South Pacific. Biol. Conserv. 116, 433–438.

Crain, D.A., Bolten, A.B., Bjorndal, K.A., 1995. Effects of beach nourishment on sea turtles: review and research initiatives. Restor. Ecol. 3, 95–104.

Cruce, J.A., 2008. Monitoring of nesting green turtles (*Chelonia mydas*) on Loosiep Island, Ulithi Atoll, Yap, Federated States of Micronesia. Draft 2008 Technical Report Prepared for JIMAR.

Cruce, J.A., 2009. Monitoring of nesting green turtles (*Chelonia mydas*) in Ulithi Atoll, Yap, Federated Sates of Micronesia. Testudo 7.

Cruz, M.G., Rodríguez-Clarkand, K., Mavarez, J., 2010. Las tortugas verdes de Isla de Aves, Venezuela, como modelo de conservacion: la oviposicion y el tamano de las hembras [WWW Document]. Proc. 30th Annu. Symp. Sea Turt. Biol. Conserv. URL http://iconferences.seaturtle.org/preview.shtml (accessed 7.6.11).

Cruz, R.D., 2002. Marine Turtle Distribution in the Philippines, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. Honolulu, HI: Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 57–66.

Cuevas, E., de los Ángeles Liceaga-Correa, M., Mariño-Tapia, I., 2010. Influence of Beach Slope and Width on Hawksbill (*Eretmochelys imbricata*) and Green Turtle (*Chelonia mydas*) Nesting Activity in El Cuyo, Yucatán, Mexico. Chelonian Conserv. Biol. 9, 262–267.

Cummings, V., 2002. Sea turtle conservation in Guam, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. February 5–8, 2002. Honolulu, HI: Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 37–38.

D'Amato, A.F., Marczwski, M., 1993. Biological aspects of marine turtles (Cheloniidae) during nesting season 1990-1991 at Praia do Forte area, district of Mata de Sao Joao, Bahia, Brazil. Arq. Biol. e Tecnol. 36, 513–519.

Da Silva Ferreira, M.B.M., 2012. Nesting habitat preferences and nest predation of green turtles (*Chelonia mydas*) in the Bijagós Archipelago, Guinea Bissau. Universidade de Lisboa.

Dalleau, M., Ciccione, S., Mortimer, J.A., Garnier, J., Benhamou, S., Bourjea, J., 2012. Nesting Phenology of Marine Turtles: Insights from a Regional Comparative Analysis on Green Turtle (*Chelonia mydas*). PLoS One 7, e46920.

Daly, T., 1990. The development of a regional sea turtle program in the South Pacific, in: Richardson, T.H., Richardson, 1.1., Donnelly, M. (Eds.), Proceedings of the Tenth Annual Workshop on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEW-278, pp. 169–172.

Daniel, R.S., Smith, K.U., 1947. The sea-approach behavior of the neonate loggerhead turtle (*Caretta caretta*). J. Comp. Physiol. Psychol. 406, 413–420.

Daniels, R.C., White, T.W., Chapman, K.K., 1993. Sea-level rise: Destruction of threatened and endangered species habitat in South Carolina. Environ. Manage. 17, 373–385.

Dattatri, S., Samarajiva, D., 1983. The status and conservation of sea turtles in Sri Lanka.

Davis, G.E., Whiting, M.C., 1977. Loggerhead sea turtle nesting in Everglades National Park, Florida, U.S.A. Herpetologica 33, 18–28.

Davis, R.W., Evans, W.E., Wursig, B.G., 2000. Cetaceans, Sea Turtles and Seabirds in the Northern Gulf of Mexico: Distribution, Abundance and Habitat Association. Volume III: data appendix, Habitat. Prepared by Texas A&M University at Galveston and the National Marine Fisheries Service. U.S. Department of the Interior, Geological Survey, Biological Resources Division, USGSBRD/CR-1999-0006 and Minerals Management Service, Gulf of Mexico OCS Region, Ne.

De Gurjao, L.M., de Freitas, J.E.P., Araujo, D.S., 2005. Observations o marine turtles during seismic surveys of Bahia, northeastern Brazil. Mar. Turt. Newsl. 108, 8–9.

De Haro, A., Harrison, E., 2007. Report on the 2006 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation and the Ministry of Envirionment, Engery and Telecommunications of Costa rica.

De Haro, A., Troëng, S., 2006. Report on the 2005 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation and the Ministry of Envirionment, Engery and Telecommunications of Costa rica.

De los Rios y Loshuertos, A.G., Moumni, A., Ocana, O., Bazairi, H., 2008. Sea turtles of Morocco: A 15 year study, in: Kalb, H., Rohde, A., Gayheart, K., Shankar, K. (Eds.), Proceedings of the Twenty-Fifth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-582, p. 170.

De Silva, G.S., 1982. The status of sea turtle populations in East Malaysia and the South China Sea, in: Bjorndal, K. (Ed.), The Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C., pp. 327–337.

Debade, X., Nolasco del Aguila, D., Harrison, E., 2008. Report on the 2007 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation and the Ministry of Envirionment, Engery and Telecommunications of Costa rica.

Defense Environmental Network and Information Exchange, n.d. Wake Island fact sheet [WWW Document]. URL http://www.denix.osd.mil/nr/crid/Coral_Reef_Iniative_Database/Wake_Atoll_files/Wake_Factsheet.pdf (accessed 4.17.13).

Deflorio, M., Aprea, A., Corriero, A., Santamaria, N., De Metrio, G., 2005. Incidental captures of sea turtles by swordfish and albacore longlines in the Ionian sea. Fish. Sci. 71, 1010–1018.

Delgado, C., Alvarado-Díaz, J., 2006. Recovery of the black sea turtle (Chelonia agassizii) in Michoacan, Mexico: An integrated conservation approach.

Delgado-Trejo, C., 2012. Recovery of the black sea turtle (Chelonia agassizi) of Michoacan, Mexico: 30 years (1982-2012) in black sea turtle conservation. U.S. Fish and Wildlife Service, Final Report 2011-2012.

Delgado-Trejo, C., Alvarado-Figueroa, J., 2012. Current Conservation Status of the Black Sea Turtle in Michoacan, Mexico, in: Seminoff, J., Wallace, B. (Eds.), Sea Turtles of the Eastern Pacific. University of Arizona Press, pp. 364–378.

Demetropoulos, A., 2000. Impact of tourism development on marine turtle nesting: strategies and actions to minimise impact, in: 20th Meeting of the Convention on the Conservation of European Wildlife and Natural Habitats. Council of Europe, T-PVS 41, Strasbourg, pp. 1–28.

Demetropoulos, A., Hadjichristophorou, M., 1989. Sea Turtle Conservation in Cyprus. Mar. Turt. Newsl. 44, 4–6.

Demetropoulos, A., Hadjichristophorou, M., 1995. Manual on Marine Turtle Conservation in the Mediterranean. UNEP(MAP)SPA/IUCN/CWS/Fisheries Department, Manre, Cyprus.

Demetropoulos, A., Hadjichristophorou, M., 2008. Conservation practices. Addendum 1 to Manual on marine turtle conservation in the Mediterranean. MANRE, Cyprus.

Demetropoulos, A., Hadjichristophorou, M., 2009. The Cyprus turtle conservation project - 29 years on, in: Demetropoulos, A., Turkozan, O. (Eds.), Proceedings of the Second Mediterranean Conference on Marine Turtles. Barcelona Convention – Bern Convention – Bonn Convention (CMS), Kemer, Turkey, pp. 19–26.

Demetropoulos, A., Hadjichristophorou, M., 2010. Cyprus - Region B, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Policies. IUCN, Gland, Switzerland, p. 53–.

Demirayak, F., 1999. The Status of the Green Turtle *Chelonia mydas* Nesting Habitat in Kazanlı, Akyatan and Samandağ on the Turkish Mediterranean Coast. United Kingdom.

Department of Environmental and Natural Resources of Puerto Rico, 2012. Status of Marine Turtles Nesting Beach Productivity in Puerto Rico. Endangered Species Program, Department of Environmental and Natural Resources of Puerto Rico (DNER) Internal Report Submitted to USFWS.

Department of Land and Natural Resources, 2013. Green turtle nesting on Maui, Department of Land and Natural Resources input on green sea turtle ESA review, personal communication.

Dermawan, A., 2002. Marine turtle management and conservation in Indonesia, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. February 5-8, 2002, Honolulu, Hawaii, USA. Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 67–76.

Dethmers, K.E.M., 2000. Demography of a Green Turtle Population Nesting on Enu Island, Aru – Indonesia, in: Kalb, H., Wibbels, T. (Eds.), Proceedings of the Nineteenth Annual Symposium on Sea Turtle Biology and Conservation. U.S. Dept. Commerce, NOAA Technical Memorandum NMFS-SEFSC-443, South Padre Island, Texas, pp. 253–254.

Dethmers, K.E.M., 2010. Ecology and phylogeography of Australasian green turtle (*Chelonia mydas*) populations; A case study for conservation from Aru, Indonesia. Ph.D. thesis. Radboud University Nijmegen, the Netherlands.

Dethmers, K.E.M., Broderick, D., Moritz, C., FitzSimmons, N.N., Limpus, C.J., Lavery, S., Whiting, S.D., Guinea, M., Prince, R.I.T., Kennett, R., 2006. The genetic structure of Australasian green turtles (*Chelonia mydas*): exploring the geographical scale of genetic exchange. Mol. Ecol. 15, 3931–3946.

Dethmers, K.E.M., Jensen, M.P., FitzSimmons, N.N., Broderick, D., Limpus, C.J., Moritz, C., 2010. Migration of green turtles (*Chelonia mydas*) from Australasian feeding grounds inferred from genetic analyses. Mar. Freshw. Res. 61, 1376–1387.

DEWHA, 2010. Australian Tuna and Billfish Longline Fisheries: Bycatch and discarding workplan.

Dirado, J.A.V., Rasado, F.V., Lopex, D.S., Vazquez, A.A., Tejedas, A.C., Viveros, M.V., 2002. Decrease of the number of nests of the green turtle (*Chelonia mydas*) in the beaches of Alto Lucero, Veracruz, Mexico, in: Mosier, A., Foley, A., Brost, B. (Eds.), Proceedings of the Twenty-Fifth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Tech. Memo. NMFS-SEFSC-477, p. 369.

DIREN, 2003. Document de prise en consideration pour le classement des iles Eparses en reserve naturelle nationale. Dir. Reg. l'Environnement la Reun. Minist. l'Ecologie du Dev. Durable, Saint Denis, La Reun.

Division of Aquatic and Wildlife Resources, 2011. Final annual progress report, Guam sea turtle recovery.

Donoso, M., Dutton, P.H., 2010. Sea turtle bycatch in the Chilean pelagic longline fishery in the southeastern Pacific: Opportunities for conservation. Biol. Conserv. 143, 2672–2684.

Dos Santos, R.G., Martins, A.S., Torezani, E., Baptistotte, C., Farias, J.N., Horta, P.A., Work, T.M., Balazs, G.H., 2010. Relationship between fibropapillomatosis and environmental quality: a case study with *Chelonia mydas* of Brazil. Dis. Aquat. Organ. 89, 87–95.

Dow, W., Eckert, K.L., Palmer, M., Kramer, P., 2007. An Atlas of Sea Turtle Nesting Habitat for the Wider Caribbean Region. The Wider Caribbean Sea Turtle Conservation Network and The Nature Conservancy. WIDECAST Technical Report No. 6, Beaufort, North Carolina.

Dryden, J., Grech, A., Moloney, J., Hamann, M., 2008. Rezoning of the Great Barrier Reef World Heritage Area: does it afford greater protection for marine turtles? Wildl. Res. 35, 477.

Duchene, S., Frey, A., Alfaro-Núñez, A., Dutton, P.H., Gilbert, M.T.P., Morin, P.A., 2012. Marine turtle mitogenome phylogenetics and evolution. Mol. Phylogenet. Evol.

Duran Najera, J.J., 1990. Nesting of three species of sea turtle in the northeast coast of the Yucatan Peninsula, Mexico, in: Richardson, T.H., Richardson, J.I., Donnelly, M. (Eds.), Proceedings of the Tenth Annual Workshop on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFC-278, Hilton Head, South Carolina, pp. 29–33.

Durmuş, S.H., 1998. An investigation biology and ecology of sea turtle population on Kazanlı and Samandağ Beaches. A Thesis submitted to the graduate school of natural and applied sciences of Dokuz Eylül University, İzmir.

Dutton, P.H., Balazs, G.H., LeRoux, R.A., Murakawa, S.K.K., Zárate, P., Martínez, L.S., 2008. Composition of Hawaiian green turtle foraging aggregations: mtDNA evidence for a distinct regional population. Endanger. species Res. 5, 37–44.

Dutton, P.H., Davis, S.K., Guerra, T., Owens, D.W., 1996. Molecular Phylogeny for Marine Turtles Based on Sequences of the ND4-Leucine tRNA and Control Regions of Mitochondrial DNA. Mol. Phylogenet. Evol. 5, 511–521.

Dutton, P.H., Frey, A., 2009. Characterization of polymorphic microsatellite markers for the green turtle (*Chelonia mydas*). Mol. Ecol. Resour. 9, 354–356.

Dutton, P.H., Jensen, M.P., Frutchey, K., Frey, A., LaCasella, E., Balazs, G.H., Cruce, J., Tagarino, A., Farman, R., Tatarata, M., 2014. Genetic stock structure of green turtle (*Chelonia mydas*) nesting populations across the Pacific islands. Pacific Sci.

Dutton, P.H., Roden, S.E., Stewart, K.R., LaCasella, E.L., Tiwari, M., Formia, A., Thomé, J.C., Livingstone, S.R., Eckert, S.A., Chacon-Chaverri, D., Rivalan, P., Allman, P., 2013. Population stock structure of leatherback turtles (*Dermochelys coriacea*) in the Atlantic revealed using mtDNA and microsatellite markers. Conserv. Genet. 14, 625–636.

Eaton, C., McMichael, E., Witherington, B.E., Foley, A., Hardy, R., Meylan, A.B., 2008. In-water sea turtle monitoring and research in Florida: review and recommendations. U.S. Dep. Commerce, NOAA Tech. Memo. NMFS-OPR-38, 233 p.

Eberdong, J., Klain, S., 2008. Palau marine turtle conservation and monitoring program, in: Rees, A.F., Frick, M., Panagopoulou, A., Williams, K. (Eds.), Proceedings of the Twenty-Seventh Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-569, pp. 230–231.

Echternacht, A.C., Burton, F.J., Blumenthal, J.M., 2011. The amphibians and reptiles of the Cayman Islands: Conservation issues in the face of invasions, in: Hailey, A., Wilson, B.S., Horrocks, J.A. (Eds.), Conservation of Caribbean Island Herpetofaunas, Vol. 1, Conservation Biology and the Wider Caribbean. Brill, Leiden, pp. 129–147.

Eckert, K.L., 1993. The biology and population status of marine turtles in the North Pacific Ocean.

Eguchi, T., Seminoff, J.A., LeRoux, R.A., Dutton, P.H., Dutton, D.L., 2010. Abundance and survival rates of green turtles in an urban environment: coexistence of humans and an endangered species. Mar. Biol. 157, 1869–1877.

Eguchi, T., Seminoff, J.A., LeRoux, R.A., Prosperi, D., Dutton, D.L., Dutton, P.H., 2012. Morphology and Growth Rates of the Green Sea Turtle (*Chelonia mydas*) in a Northern-most Temperate Foraging Ground. Herpetologica 68, 76–87.

Ehrhart, L.M., 1983. Marine turtles of the Indian River lagoon system. Florida Sci. 46, 337–346.

Ehrhart, L.M., Redfoot, W.E., Bagley, D.A., 2007. Marine turtles of the central region of the Indian River Lagoon System, Florida. Florida Sci. 70, 415–434.

Eisentraut, M., 1964. Meeresschildkroten an der Kuste von Fernando Poo. Nat. Mus. 94, 471–475.

Ekanayake, E.M.L., Kapurusinghe, T., Premakumara, M.G.C., Saman, M.M., Rathnakumara, A.M.D.S., Rajakaruna, R.S., Samaraweera, P., Ranawana, K.B., 2013. Re-nesting movements and post-nesting migrations of green turtles tagged in two turtle rookeries in Sri Lanka, in: Blumenthal, J.M., Panagopoulou, A., Rees, A.F. (Eds.), Proceedings of the Thirtieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-640, Goa, India, p. 123.

Ekanayake, E.M.L., Kapurusinghe, T., Saman, M.M., Rathnakumara, A.M.D.S., 2011. A declining trend of nesting frequency of sea turtles at the largest rookery in Sri Lanka, in: Jones, T.T., Wallace, B.P. (Eds.), Proceedings of the 31st Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-631, San Diego, California, p. 216.

Ekanayake, E.M.L., Rajakaruna, R.S., Kapurusinghe, T., Saman, M.M., Rathnakumara, D.S., Samaraweera, P., Ranawana, K.B., 2010. Nesting behaviour of the green turtle at Kosgoda rookery, Sri Lanka. Ceylon J. Sci. 39, 109–120.

Ekanayake, E.M.L., Ranawana, K.B., 2001a. The average egg count for nesting sea turtles on Rekawa beach in Sri Lanka, in: Abstracts of the Fourth World Congress of Herpetology 3rd – 9th December 2001. Bentota, Sri Lanka, pp. 29–30.

Ekanayake, E.M.L., Ranawana, K.B., 2001b. Estimation of Sex Ratio of green turtle hatchlings at Rekawa turtle rookery in Sri Lanka., in: Abstracts of the Fourth World Congress of Herpetology, 3rd –9th December 2001. Bentota, Sri Lanka, pp. 28–29.

Ekanayake, E.M.L., Ranawana, K.B., Kapurusinghe, T., 2004. Estimation of the average number of nests for green turtle, on the Rekawa beach in Southern Sri Lanka. Three-year study from September 1996 to September 1999, in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings of the Twenty-First Annual Syumposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMSF-SEFSC-528, Philadelphia, PA, pp. 160–161.

Ekanayake, E.M.L., Ranawana, K.B., Kapurusinghe, T., Premakumara, M.G.C., Saman, M.M., 2002. Marine turtle conservation in Rekawa turtle rookery in southern Sri Lanka. Ceylon J. Sci. 30, 79–88.

El Shaer, H., L. Samaha, L., Jaradi, G., 2012. Lebanon's marine protected area strategy supporting the management of important marine habitats and species in Lebanon.

Eliazar, P.J., Bjorndal, K.A., Bolten, A.B., 2000. Early report of fibropapilloma from St. Croix, USVI. Mar. Turt. Newsl. 89, 16.

Elmaz, Ç., 2005. Mersin-Kazanlı Kumsalı'nda yuvalayan *Chelonia mydas* ve *Caretta caretta* (Linneaus 1758) türü deniz kaplumbağaları populasyonlarının durumu ve korunması. Mersin Üniversitesi Fen Bilimleri Enstitüsü Su Ürünleri ABD.

El-Sadek, I., Mancini, A., Hanafy, M.H., Girondot, M., 2013. Green turtle nesting activities on Zabargard Island, a major rookery in the southern Egyptian Red Sea, in: Tucker, T., Belskis, L., Panagopoulou, A., Rees, A.F., Frick, M., Williams, K., LeRoux, R., Stewart, K. (Eds.), Proceedings of the Thirty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NOAA NMFS-SEFSC-645, Baltimore, Maryland, pp. 174–175.

Elschner, C., 1915. The leeward islands of the Hawaiian group; Contributions to the knowledge of the Islands of Oceania. Sunday Advertiser, Honolulu.

Emanuel, K., 2005. Genesis and maintenance of "Mediterranean hurricanes". Adv. Geosci. 2, 217–220.

Emory, K.P., 1975. Material culture of the Tuamotu Archipelago. Department of anthropology, Bishop Museum Press, Honolulu.

Encalada, S.E., Lahanas, P.N., Bjorndal, K.A., Bolker, B., Miyamoto, M.M., Bowen, B.W., 1996. Phylogeography and population structure of the Atlantic and Mediterranean green turtle *Chelonia mydas*: a mitochondrial DNA control region sequence assessment. Mol. Ecol. 5, 473–483.

Ene, A.R., Su, M., Lemaire, S., Rose, C., Schaff, S., Moretti, R., Lenz, J., Herbst, L.H., 2005. Distribution of chelonid fibropapillomatosis-associated herpesvirus variants in Florida: molecular genetic evidence for infection of turtles following recruitment to neritic developmental habitats. J. Wildl. Dis. 41, 489–497.

Engeman, R.M., Martin, R.E., Smith, H.T., Woolard, J., Crady, C.K., Shwiff, S. a., Constantin, B., Stahl, M., Griner, J., 2005. Dramatic reduction in predation on marine turtle nests through improved predator monitoring and management. Oryx 39, 318–326.

Environment Australia, 2003. Recovery Plan for Marine Turtles in Australia.

Epperly, S.P., 2003. Fisheries-related mortality and turtle excluder devices (TEDs), in: Lutz, P.L., Musick, J.A., Wyneken, J. (Eds.), The Biology of Sea Turtles, Volume II. CRC Press, Boca Raton, Florida, pp. 339–354.

Epperly, S.P., Avens, L.A., Garrison, L., Henwood, T., Hoggard, W., Mitchell, J., Nance, J., Poffenberger, J., Sasso, C., Scott-Denton, E., Yeung, C., 2002. Analysis of sea turtle bycatch in the commercial shrimp fisheries of southeast U.S. waters and the Gulf of Mexico. U.S. Department of Commerce, NOAA Technical Memorandum NMFS- SEFSC-490, 88 pp.

Epperly, S.P., Braun-McNeill, J., Richards, P., 2007. Trends in catch rates of sea turtles in North Carolina, USA. Endanger. Species Res. 3, 283–293.

Epperly, S.P., Teas, W.G., 2002. Turtle excluder devices - Are the escape openings large enough? Fish. Bull. 100, 466–474.

Ernest, R.G., Martin, R.E., 1999. Martin County Beach Nourishment Project: Sea Turtle Monitoring and Studies 1997 Annual Report and Final Assessment, Construction. Jensen Beach, Florida.

Evans, K.E., Vargas, A.E., 1998. Sea turtle egg commercialization in Isla de Canas, Panama, in: Byles, R., Fernandez, Y. (Eds.), Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-412, p. 158.

Farrell, J.W., Raffi, I., Janacek, T.R., Murray, D.W., Levitan, M., Dadey, K.A., Emeis, K.-C., Lyle, M., Flores, J.A., Hovan, S., 1995. Late Neogene sedimentation patterns in the eastern equatorial Pacific Ocean., in: Pisias, N.G. (Ed.), Proc. Ocean Drill. Program Sci. Results, vol.138,. Ocean Drill. Program, College Station, Texas, pp. 717–756.

Fatima, E., Andrews, H. V., John, J., Shanker, K., 2011. Status of marine turtles in Cuthbert Bay, Middle Andaman Islands. Mar. Turt. Newsl. 130, 6–9.

Fennessy, S.T., Isaksen, B., 2007. Can bycatch reduction devices be implemented successfully on prawn trawlers in the Western Indian Ocean? African J. Mar. Sci. 29, 453–463.

Ferreira, B., Garcia, M., Jupp, B.P., Al-Kiyumi, a., 2006. Diet of the Green Turtle (*Chelonia mydas*) at Ra's Al Hadd, Sultanate of Oman. Chelonian Conserv. Biol. 5, 141.

Fiedler, F.N., Sales, G., Giffoni, B.B., Monteiro-Filho, E.L.A., Secchi, E.R., Bugoni, L., 2012. Driftnet fishery threats sea turtles in the Atlantic Ocean. Biodivers. Conserv. 21, 915–931.

Figueroa, A., Alvarado, J., Hernandez, F., Rodriguez, G., Robles, J., 1993. Population recovery of the sea turtles of Michoacan, Mexico: an integrated conservation approach. Final Rep. 1991-1992 Submitt. to World Wildl. Fund-U.S. U.S. Fish Wildl. Serv.

Firdous, F., 2001. Sea turtle conservation and educationin Karachi, Pakistan. ASEAN Rev. Biodivers. Environ. Conserv. 1–10.

Firouz, E., 2005. The complete fauna of Iran. I.B. Tauris & Co. Ltd, London, U.K.

Fish, M.R., Côté, I.M., Gill, J.A., Jones, A.P., Renshoff, S., Watkinson, A.R., 2005. Predicting the impact of sea-level rise on Caribbean sea turtle nesting habitat. Conserv. Biol. 19, 482–491.

Fish, M.R., Côté, I.M., Horrocks, J.A., Mulligan, B., Watkinson, A.R., Jones, A.P., 2008. Construction setback regulations and sea-level rise: mitigating sea turtle nesting beach loss. Ocean Coast. Manag. 51, 330–341.

Fisher, C., 2012. Nesting population of sea turtles in four main beaches of eastern Tioman Island, Malaysia. Juara Turtle Project.

Fitzgerald, D.B., Ordway, E., Honarvar, S., Hearn, G.W., 2011. Challenges Confronting Sea Turtle Conservation on Bioko Island, Equatorial Guinea. Chelonian Conserv. Biol. 10, 177–180.

FitzSimmons, N.N., Moritz, C., Limpus, C.J., Pope, L., Prince, R.I.T., 1997. Geographic structure of the mitochondrial and nuclear gene polymorphisms in Australian green turtle populations and male-biased gene flow. Genetics 147, 1843–1854.

Flaxman, M., Vargas-Moreno, J.C., 2011. Considering Climate Change in State Wildlife Action Planning: A Spatial Resilience Planning Approach. Cambridge MA. Research Report FWC-2011. Dept of Urban Studies and Planning, Massachusetts Institute of Technology.

Fleming, E.H., 2001. Swimming Against the Tide: Recent Surveys of Exploitation, Trade, And Management of Marine Turtles In the Northern Caribbean, America. Washington, D.C., USA.

Flores, L., De Ben, A., Salado, I., Morote, E., Clavero, J., Yllescas, L., 2006. Relationships between marine turtles and communities on the coast of Mauritania, in: Pilcher, N.J. (Ed.), Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-536, p. 191.

Florida Fish and Wildlife Conservation Commission, 2012a. Green turtle nesting in Florida [WWW Document]. URL http://myfwc.com/research/wildlife/sea-turtles/nesting/green-turtle/ (accessed 11.7.12).

Florida Fish and Wildlife Conservation Commission, 2012b. Index nesting beach survey totals (1989-2012) [WWW Document]. URL http://myfwc.com/research/wildlife/sea-turtles/nesting/beach-survey-totals/ (accessed 11.7.12).

Florida Fish and Wildlife Conservation Commission, 2013. Florida statewide total nest numbers from 1979-2012. Personal communication with Dean Bagley.

Foley, A., Singel, K., Dutton, P.H., Summer, T., Redlow, A.E., Lessman, J., 2007. Characteristics of a Green Turtle (*Chelonia mydas*) Assemblage in Northwestern Florida Determined During a Hypothermic Stunning Event 2007, 131–143.

Foley, A.M., Schroeder, B.A., Redlow, A.E., Fick-Child, K.J., Teas, W.G., 2005. Fibropapillomatosis in stranded green turtles (*Chelonia mydas*) from the eastern United States (1980-98): Trends and associations with environmental factors. J. Wildl. Dis. 41, 29–41.

Fonseca, L.G., Villachica, W.N., Matarrita, R.E., Valverde, R.A., 2011. Reporete final de anidacion de tortuga verde (*Chelonia mydas*), Playa Nancite, Parque Nacional Santa Rosa, Costa Rica (Temporada 2010-2011).

Food and Agriculture Organization of the United Nations, 2001. Fishery and aquaculture country profiles: Qatar [WWW Document]. URL http://www.fao.org/fishery/countrysector/FI-CP_QA/en (accessed 5.7.13).

Food and Agriculture Organization of the United Nations, 2004. Expert consultation on interactions between sea turtles and fisheries within an ecosystem context. FAO Fish. Rep. No. 738.

Formia, A., 1999. Les tortues marines de la Baie de Corisco. Canopee 14, i–ii.

Formia, A., Broderick, A.C., Glen, F., Godley, B.J., Hays, G.C., Bruford, M.W., 2007. Genetic composition of the Ascension Island green turtle rookery based on mitochondrial DNA: implications for sampling and diversity. Endanger. Species Res. 3, 145–158.

Formia, A., Godley, B.J., Dontaine, J.-F., Bruford, M.W., 2006. Mitochondrial DNA diversity and phylogeography of endangered green turtle (*Chelonia mydas*) populations in Africa. Conserv. Genet. 7, 353–369.

Formia, A., Tiwari, M., Fretey, J., Billes, A., 2003. Sea turtle conservation along the Atlantic Coast of Africa. Mar. Turt. Newsl. 100, 33–37.

Fosberg, F.R., 1990. A review of the natural history of the Marshall Islands. Atoll Res. Bull. 330, 99 pages.

Fowler, L.E., 1979. Hatching Success and Nest Predation in the Green Sea Turtle, *Chelonia mydas*, at Tortuguero, Costa Rica. Ecology 60, 946–955.

Fraizer, J., 1971. Observations on sea turtles at Aldabra Atoll. Philos Trans. R. Soc. Lond. B. Biol. Sci. 260, 373–410.

Francia, G., 2011. Numero de nidos protegidos in situ y en tortugario en Playa Junquillal, desde diciembre del 2004 hasta abril de 2011. Unpublished Report, WWF / Asociacion Vida Verdiazul.

Francke, D.L., 2013. Marine Turtle Strandings in the Hawaiian Islands January-December 2012. Marine Turtle Research Program, Protected Species Division, NOAA Pacific Islands Fisheries Science Center.

Francke, D.L., 2014. Marine turtle strandings in the Hawaiian Islands January-December 2013. Marine Turtle Research Program, Protected Species Division, NOAA Pacific Islands Fisheries Science Center.

Francke, D.L., Hargrove, S.A., Vetter, W., Winn, C.D., Balazs, G.H., Hyrenbach, K.D., 2013. Behavior of juvenile green turtles in a coastal neritic habitat: validating time-depth-temperature records using visual observations. J. Exp. Mar. Bio. Ecol. 444, 55–65.

Francour, P., Ganteaume, A., Poulain, M., 1999. Effects of boat anchoring in Posidonia oceanica seagrass beds in the Port-Cros National Park (north-western Mediterranean Sea). Aquat. Conserv. Mar. Freshw. Ecosyst. 9, 391–400.

Frazer, N.B., Ehrhart, L.M., 1985. Preliminary growth models for green, *Chelonia mydas*, and loggerhead, *Caretta caretta*, turtles in the wild. Copeia 1, 73–79.

Frazer, N.B., Ladner, R.C., 1986. A growth curve for green sea turtles, *Chelonia mydas*, in the U.S. Virgin Islands, 1913-14. Copeia 3, 798–802.

Frazier, J.G., 1975. Marine turtles of the Western Indian Ocean. Oryx 13, 164–175.

Frazier, J.G., 1980. Exploitation of Marine Turtles in the Indian Ocean. Hum. Ecol. 8, 329–370.

Frazier, J.G., 1982. The status of marine turtles in the central western Indian Ocean, in: Bjorndal, K.A. (Ed.), The Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C., pp. 385–390.

Frazier, J.G., 1985. Marine turtles in the Comoro Archipelago. North-Holland Publishing Company, New York.

Frazier, J.G., 1990. Biology and conservation of the sea turtles in the Indian Ocean, in: Daniel, J.C., J.S., S. (Eds.), Conservation in Developing Countries: Problems and Prospects. Oxford University Press, Oxford, England, pp. 364–386.

Frazier, J.G., 2012. Critical tools for conserving marine turtles in the Eastern Pacific Ocean, in: Seminoff, J., Wallace, B. (Eds.), Sea Turtles of the Eastern Pacific. University of Arizona Press, Tucson, Arizona, pp. 154–190.

Fretey, J., 1984. The national report for the country of French Guiana, in: Bacon, P., Berry, F., Bjorndal, K.A., Hirth, H., Ogren, L., Weber, M. (Eds.), Proceedings of the Western Atlantic Turtle Symposium. Univ. Miami Press, pp. 177–183.

Fretey, J., 2001. Biogeography and Conservation of Marine Turtles of the Atlantic Coast of Africa. Convention of Migratory Species (CMS) Technical Series Publication No. 6.

Fretey, J., Fourmy, J., 1996. The status of sea turtle conservation in French Territories of the Indian Ocean: Mayotte, in: Status of Sea Turtle Conservation in the Western Indian Ocean. Proceedings of the Western Indian Ocean Training Workshop and Strategic Planning Sesson on Sea Turtles Held at Sodwana Bay, South Africa. UNEP Regional Seas Reports and Studies No. 165, pp. 133–143.

Fretey, J., Hama, L.F., 2012. Knowledge and conservation of loggerheads and green turtles on the coast of Mauritania. U.S. Fish and Wildlife Service and Foundation Nature & Decouvertes.

Fretey, J., Lescure, J., 1992. New protection for sea turtles in France. Mar. Turt. Newsl. 57, 11–12.

Fretey, J., Malaussena, P., 1991. Sea turtle nesting in Sierra Leone, West Africa. Mar. Turt. Newsl. 54, 10–12.

Frey, A., Dutton, P.H., Balazs, G.H., 2013. Insights on the demography of cryptic nesting by green turtles (*Chelonia mydas*) in the main Hawaiian Islands from genetic relatedness analysis. J. Exp. Mar. Bio. Ecol. 442, 80–87.

Friedlander, A.M., Aeby, G., Brainard, R., Brown, E., Chaston, K., Clark, A., Mcgowan, P., Montgomery, T., Walsh, W., Williams, I., Wiltse, W., 2008. The state of coral reef ecosystems of the Main Hawaiian Islands, in: Waddell, J., Clarke, A.M. (Eds.), The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States: 2008. NOAA Technical Memorandum NOS NCCOS 73. NOAA/NCCOS Center for Coastal Monitoring and Assessment's Biogeography Team. Silver Spring, MD, pp. 219–253.

Friedlander, A.M., Aeby, G., Brown, E., Clark, A., Coles, S., Dollar, S., Hunter, C., Jokiel, P., Smith, J., Walsh, B., Williams, I., Wiltse, W., 2005. The State of Coral Reef Ecosystems of the Main Hawaiian Islands, in: The State of Coral Reef Ecosystems of the United States and Pacific Freely Associated States: 2005. NOAA Technical Memorandum NOS NCCOS 11. NOAA/NCCOS Center for Coastal Monitoring and Assessment's Biogeography Team. Silver Spring, MD. 522 pp, pp. 222–269.

Friedlander, A.M., Brown, E.K., Monaco, M.E., Clark, A., 2006. Fish Habitat Utilization Patterns and Evaluation of the Efficacy of Marine Protected Areas in Hawaii: Integration of NOAA Digital Benthic Habitat Mapping and Coral Reef Ecological Studies. NOAA Technical Memorandum NOS NCCOS 23. Silver Spring, MD. 217 pp.

Fritts, T.H., McGehee, M.A., 1982. Effects of petroleum on the development and survival of marine turtle embryos. U.S. Fish Wildl. Serv. Rep. USFWS/OBS-82/37. 41 pages.

Fuentes, M.M.P.B., Dawson, J., Smithers, S., Hamann, M., Limpus, C.J., 2010a. Sedimentological characteristics of key sea turtle rookeries: potential implications under projected climate change. Mar. Freshw. Res. 61, 464–473.

Fuentes, M.M.P.B., Hamann, M., Limpus, C.J., 2010b. Potential impacts of projected sea level rise to sea turtle rookeries. Aquat. Conserv. Mar. Freshw. Ecosyst.

Fuentes, M.M.P.B., Limpus, C.J., Hamann, M., 2011. Vulnerability of sea turtle nesting grounds to climate change. Glob. Chang. Biol. 17, 140–153.

Fuller, J.E., Eckert, K.L., Richardson, J.I., 1992. WIDECAST Sea turtle recovery action Plan for Antigua and Barbuda (Karen L. Eckert, Editor). CEP Technical Report No. 16 UNEP Caribbean Environment Programme, Kingston, Jamaica. 88 pages.

Fuller, W.J., Broderick, A.C., Enever, R., Thorne, P., Godley, B.J., 2010a. Motile homes: a comparison of the spatial distribution of epibiont communities on Mediterranean sea turtles. J. Nat. Hist. 44, 1743–1753.

Fuller, W.J., Broderick, A.C., Glen, F., Kusetogulları, H., Godley, B.J., 2010b. Cyprus - Region A, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Policies. IUCN, Gland, Switzerland, pp. 41–52.

Gaertner, M.A., Jacob, D., Gil, V., Domínguez, M., Padorno, E., Sánchez, E., Castro, M., 2007. Tropical cyclones over the Mediterranean Sea in climate change simulations. Geophys. Res. Lett. 34, L14711.

Galair, A.S., 2009. RMCO sheds light on turtle harvest in Somalia [WWW Document]. Indian Ocean - South-East Asian Mar. Turt. Memo. Underst. Profile Mon. URL http://www.ioseaturtles.org/pom_detail.php?id=86

Gardner, S.C., Nichols, W.J., 2001. Assessment of Sea Turtle Mortality Rates in the Bahía Magdalena Region , Baja California Sur , México. System 4, 197–199.

Garduño-Andrade, M., Maldonado, A., M‡arquez, R., Schroeder, B.A., Balazs, G.H., 2000. Satellite tracking of an adult male and female green turtle from Yucatan in the Gulf of Mexico, in: Kalb, H., Wibbles, T. (Eds.), Proceedings of the Nineteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Tech. Memo. NMFS-SEFSC-343, Miami, FL, p. 158.

Garland, K.A., Carthy, R.R., 2010. Changing taste preferences, market demands and traditions in Pearl Lagoon, Nicaragua: a community reliant on green turtles for income and nutrition. Conserv. Soc. 8, 55–72.

Garnier, J., Hill, N., Guissamulo, A., Silva, I., Witt, M.J., Godley, B.J., 2012. Status and community-based conservation of marine turtles in the northern Querimbas Islands (Mozambique). Oryx 46, 359–367.

Gavilan, F.G.M., Andreu, G.N., 1998. The green turtle (*Chelonia mydas*) in Cuba, in: Epperly, S.P., Braun, J. (Eds.), Proceedings of the Seventeenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-415, 294 pages, pp. 57–59.

Gavilan, F.G.M., Font, L., Morales, E., Escobar, E., Nodarse, G., Valle, S., Garcia, J., Meneses, A., 2003. Bycatch of marine turtles in Cuban shelf-waters, in: Seminoff, J.A. (Ed.), Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503, 308 pages, pp. 8–9.

George, R.H., 1997. Health problems and diseases of sea turtles, in: Lutz, P.L., Musick, J.A. (Eds.), The Biology of Sea Turtles. CRC Press, Boca Raton, Florida, pp. 363–409.

Gianguzza, P., Russo, G., Violani, C., Zava, B., 2000. Ascertined record of the green sea turtle, *Chelonia mydas* (L., 1758) in the Tyrrhenian Sea (Testudinata, Cheloniidae). Atti della Società italiana di scienze naturali e del Museo civico di storia naturale di Milano 141, 19–22.

Gilbert, P.M., Anderson, D.M., Gentien, P., Graneli, E., Sellner, K.G., 2005. The global complex phenomena of harmful algal blooms. Oceanography 18, 136–147.

Girard, A., NDembé, H., Bréheret, N., 2013. Fibropapillomas in green turtles along the coast of the Congo-Brazzaville. Seven years of observations give an insight into a rising issue in central Africa, in: Proceedings of the 33rd Annual Symposium on Sea Turtle Biology and Conservation. Baltimore, Maryland.

Girard, C., Sudre, J., Benhamou, S., Roos, D., Luschi, P., 2006. Homing in green turtles *Chelonia mydas*: oceanic currents act as a constraint rather than as an information source. Mar. Ecol. Prog. Ser. 322, 281–289.

Giron, L.E., 2006. Legislation that protects sea turtles in Guatemala, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Book of Abstracts - 26th Annual Symposium on Sea Turtle Biology and Conservation. Creta, Greece, pp. 121–122.

Glen, F., Broderick, A.C., Godley, B.J., Hays, G.C., 2005. Patterns in the emergence of green (*Chelonia mydas*) and loggerhead (*Caretta caretta*) turtle hatchlings from their nests. Mar. Biol. 146, 1039–1049.

Glen, F., Broderick, A.C., Godley, B.J., Hays, G.C., 2006a. Rhythmic Throat Oscillations in Nesting Green Turtles (*Chelonia mydas*). Chelonian Conserv. Biol. 5, 299–301.

Glen, F., Broderick, A.C., Godley, B.J., Hays, G.C., 2006b. Thermal control of hatchling emergence patterns in marine turtles. J. Exp. Mar. Bio. Ecol. 334, 31–42.

Glen, F., Mrosovsky, N., 2004. Antigua revisited: the impact of climate change on sand and nest temperature at a hawksbill turtle (Eretmochelys imbircata) nesting beach. Glob. Chang. Biol. 10, 2036–2045.

Godfrey, M.H., Mrosovsky, N., 2006. Pivotal temperature for green sea turtles, *Chelonia mydas*, nesting in Suriname. Herpetol. J. 16, 55–61.

Godfrey, M.H., Mrosovsky, N., Barreto, R., 1996. Estimating past and present sex ratios of sea turtles in Suriname. Can. J. Zool. 74, 267–277.

Godley, B.J., Almeida, A.P., Barbosa, C., Broderick, A.C., Catry, P., Hays, G.C., Indjai, B., 2003a. Using satellite telemetry to determine migratory corridors and foraging grounds of green turtles nesting at Poilão, Guinea Bissau: Report to project donors.

Godley, B.J., Broderick, A.C., Campbell, L.M., Ranger, S., Richardson, P.B., 2004. Towards a Molecular Profile of Marine Turtles in the Caribbean Overseas Territories. Final project report for the Department of Environment, Food and Rural Affairs and the Foreign and Commonwealth Office, pp. 223–236.

Godley, B.J., Broderick, A.C., Glen, F., Hays, G.C., 2002a. Temperature-dependent sex determination of Ascension Island green turtles. Mar. Ecol. Prog. Ser. 226, 115–124.

Godley, B.J., Broderick, A.C., Mrosovsky, N., 2001. Estimating hatchling sex ratios of loggerhead turtles in Cyprus from incubation duration. Mar. Ecol. Prog. Ser. 210, 195–201.

Godley, B.J., Furness, R.W., Solomon, S.E., 1998a. Patterns of mortality in marine turtles in the eastern Mediterranean, in: Byles, R., Fernandez, Y. (Eds.), Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS- SEFSC-412, pp. 59–61.

Godley, B.J., Gucu, A.C., Broderick, A.C., Furness, R.W., Solomon, S.E., 1998b. Interaction between marine turtles and artisanal fisheries in the eastern Mediterranean: a probable cause for concern? Zool. Middle East 16, 49–64 ST – Interaction between marine turtles and.

Godley, B.J., Lima, E.H.S.M., Åkesson, S., Broderick, A.C., Glen, F., Godfrey, M.H., Luschi, P., Hays, G.C., 2003b. Movement patterns of green turtles in Brazilian coastal waters described by satellite tracking and flipper tagging. Mar. Ecol. Prog. Ser. 253, 279–288.

Godley, B.J., Richardson, S., Broderick, A.C., Coyne, M.S., Glen, F., Hays, G.C., 2002b. Long-term satellite telemetry of the movements and habitat utilisation by green turtles in the Mediterranean. Ecography (Cop.). 25, 352–362.

Godley, B.J., Thompson, D.R., Furness, R.W., 1999. Do heavy metal concentrations pose a threat to marine turtles from the Mediterranean Sea? Mar. Pollut. Bull. 38, 497–502.

Godoy, D.A., FitzSimmons, N.N., Boyle, M.C., 2012. A long way from home: First confirmed records of the east Pacific green turtle (*Chelonia mydas* Agassizii) in New Zealand, in: Proceedings of the Thirty-Second Annual Symposium on Sea Turtle Biology and Conservation. Oaxaca, Mexico.

Goitom, M., Weldeyohannes, S., Mengstu, T., Teclemariam, Y., 2006. Marine turtles of Eritrea, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece, p. 134.

González Carman, V., Álvarez, K.C., Prosdocimi, L., Inchaurraga, M.C., Dellacasa, R.F., Faiella, A., Echenique, C., González, R., Andrejuk, J., Mianzan, H.W., Campagna, C., Albareda, D. a., 2011. Argentinian coastal waters: A temperate habitat for three species of threatened sea turtles. Mar. Biol. Res. 7, 500–508.

Gonzalez Prieto, C., Harrison, E., 2012. Report on the 2011 green turtle program at Tortuguero, Costa Rica. Sea Turtle Conservancy and the Ministry of Envirionment, Engery and Telecommunications of Costa Rica. 19 April 2012. 62 pp.

Good, T.P., Waples, R.S., Adams, P., 2005. Updated status of federally listed ESUs of West Coast salmon and steelhead. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-66.

Goodwin, M.M., 1971. Some aspects and problems of the use and exploitation of marine turtles, Marine Turtles.

Gordon, A., Pople, A., Ng, J., 1998. Trace metal concentrations in livers and kidneys of sea turtles from south-eastern Queensland, Australia. Mar. Freshw. Res. 49, 409–414.

Gordon, A.L., Ma, S., Olson, D.B., Hacker, P., Ffield, A., Talley, L.D., Wilson, D., Baringer, M., 1997. Advection and diffusion of Indonesian Throughflow Water within the Indian Ocean South Equatorial Current. Geophys. Res. Lett. 24, 2573.

Goshe, L.R., Avens, L., Scharf, F.S., Southwood, A.L., 2010. Estimation of age at maturation and growth of Atlantic green turtles (*Chelonia mydas*) using skeletochronology. Mar. Biol. 157, 1725–1740.

Gouin, P., Petit, M., 2010. 1er Symposium international sur les tortues marines en Polynésie française. Association Te mana o te moana, Moorea, French Polynesia.

Gourlay, T., Markovina, K., Prophet, M., Whiting, A.U., 2010. Ningaloo Turtle Program Annual Report 2009-2010, Program. Exmouth, Western Australia.

Goutenègre, S., Perrin, E., Frutchey, K., 2011. Green sea turtle scoping survey on Mopelia atoll/French Polynesia.

Government of Samoa, 1998. National report to the Convention on Biological Diversity. Division of Environment and Conservation, Department of Lands Survey and Environment, December.

Government of the Federated States of Micronesia, 2004. Federated States of Micronesia's strategic development plan 2004-2023. The next 20 years: achieving economic growth & self-reliance. Vol I: policies and strategies for development. Unpublished report. 479 pages.

Graeffe, E., 1873. Samoa oder die Schifferinseln I. Topographie von Samoa. J. des Museums Godeffroy 1, 1–32.

Grant, G.S., Craig, P., Balazs, G.H., 1997. Notes on juvenile hawksbill and green turtles in American Samoa. Pacific Sci. 51, 48–53.

Green, D., 1983. Galapagos sea turtles. Not. Galapagos 38, 22–25.

Green, D., 1984. Long-Distance Movements of Galapagos Green Turtles. J. Herpetol. 18, 121.

Green, D., 1993. Growth Rates of Wild Immature Green Turtles in the Galapagos Islands, Ecuador. J. Herpetol. 27, 338.

Green, D., 1998. Basking in Galapagos green turtles., in: Epperly, S.P., Braun, J. (Eds.), Proceedings of the Seventeenth Annual Sea Turtle Symposium. NOAA Technical Memorandum NMFS-SEFSC-415, pp. 60–62.

Green, D., Ortiz-Crespo, F., 1995. Status of sea turtle populations in the central eastern Pacific, in: Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington, D.C., pp. 221–234.

Green, S.O., Webber, D.F., 2003. The effects of varying levels of eutrophication on phytoplankton and seagrass (Thalassia testudinum) populations of the Southeast coast of Jamaica. Bull. Mar. Sci. 73, 443–456.

Greenblatt, R.J., Work, T.M., Dutton, P.H., Sutton, C.A., Sprakter, T.R., Casey, R.N., Diez, C.E., Parker, D., St. Ledger, J.A., Balazs, G.H., Casey, J.W., 2005. Geographic varation in marine turtle fibropapillomatosis. J. Zoo Wildl. Med. 36, 527–530.

Greenway, M., 1977. The production and utilization of Thalassia testudinum (Konig) in Kingston Harbour. Univ. of the West Indies, Mona.

Gribble, N.A., McPherson, G., Lane, B., 1998. Effect of the Queensland Shark Control Program on non-target species: whale, dugong, turtle and dolphin, a review. Mar. Freshw. Res. 49, 645–651.

Grobler, H.J., Erzini, K., Said, A.S.B., Hamriy, S. Al, 2001. Environment of the Nesting and Feeding Grounds for Endangered Turtle Species in Dhofar (Southern Oman). Environment 151–159.

Groombridge, B., Luxmoore, R., 1989. The green turtle and hawksbill (Reptilia: Cheloniidae): world status, exploitation and trade, Flora. CITES, Lausanne, Switzerland.

Groombridge, B., Marker Kabraji, A., Latif Rao, A., 1988. Marine turtle in Baluchistan (Pakistan). Mar. Turt. Newsl. 42, 1–3.

Guam Division of Aquatic and Wildlife Resources, 2011. Final annual progress report, Guam sea turtle recovery, grant number NA10NMF4540385. Guam Department of Agriculture, Division of Aquatic and Wildlife Resources, Guam.

Guam Division of Aquatic and Wildlife Resources, 2012. Semi-annual progress report, Guam sea turtle recovery, grant number NA11NMF4540237.

Guebert-Bartholo, F., Barletta, M., Costa, M., Monteiro-Filho, E.L.A., 2011. Using gut contents to assess foraging patterns of juvenile green turtles *Chelonia mydas* in the Paranaguá Estuary, Brazil. Endanger. Species Res. 13, 131–143.

Guilbeaux, M.D., 2001. USE Relating to sea turtles, their management, and policy in the Republic of Palau: an assessment of stakeholder perception. Volumes 1 & 2. The Palau Conservation Society. 164 pages, Koror, Palau.

Guseman, J.L., Ehrhart, L.M., 1990. Green turtles on sabellariid worm reefs: initial results from studies on the Florida Atlantic coast., in: Richardson, T.H., Richardson, J.I., Donnelly, M. (Eds.), Proceedings of the Tenth Annual Workshop on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFC-278, pp. 125–127.

Gustafson, R.G., Drake, J., Ford, M.J., Myers, J.M., Holmes, E.E., Waples, R.S., 2006. Status review of Cherry Point Pacific Herring (Clupea pallasii) and Updated Status Review of the Georgia Basin Pacific Herring Distinct Population Segment under the Endangered Species Act. US Dept. Commer., NOAA Tech. Memo., NMFS-NWFSC-76 182.

Guzmán-Hernández, V., 1996. Reseña de la temporada de anidación en Campeche, México, durante 1996, Inédito.

Guzmán-Hernández, V., 1997. Informe técnico del Programa de Investigación y Protección de Tortugas Marinas en Isla Aguada, Campeche. Temporada 1997, Inédito.

Guzmán-Hernández, V., 2000. Evaluación de las poblaciones de tortugas marinas de Campeche Avances y Perspectivas. Informe Técnico de Investigación 2000 Clave:INE/DGVS/TM/007-CAMP.

Guzmán-Hernández, V., 2001. Evaluación de las poblaciones de tortugas marinas de Campeche. Sinopsis de la Protección e Investigación 1977-2001, con Reporte de Investigación 2001/INE/DGVS/TM-007-CAMP.

Guzmán-Hernández, V., 2002. Reseña de la temporada de anidación de tortugas marinas en Campeche, México; durante 2002. Con Reporte del campamento tortuguero de Isla Aguada en 2002 INE/DGVS/TM-007-CAMP.

Guzmán-Hernández, V., 2003. Temporada de anidación 2003 de tortugas marinas en Isla Aguada, Campeche, México y panorama estatal. INE/DGVS/TM-007- Camp.

Guzmán-Hernández, V., 2005. Informe final de la temporada de anidación 2004 de tortugas marinas en Isla Aguada, Campeche, Mexico. INE/DGVS/TM-007-CAMP.

Guzmán-Hernández, V., 2006a. Informe tecnico final 2006 del programa conservación de tortugas marinas en Campeche, Mexico. CONANP/LAGUNA DE TÉRMINOS.

Guzmán-Hernández, V., 2006b. Evaluación de las poblaciones de tortugas marinas de Campeche. Sinopsis de la Protección e Investigación 1977-2001, con Reporte de Investigación 2001/INE/DGVS/TM-007-CAMP, Conservación de las Tortugas Marinas de Campeche.

Guzmán-Hernández, V., 2006c. Informe técnico final del programa de conservación de tortugas marinas de Campeche, Mexico en 2005. Incluye informe del campamento Isla Aguada, INE/DGVS/TM-007-CAMP y Xicalango-Victoria.

Guzmán-Hernández, V., García Alvarado, P.A., 2009. Informe téchnico 2008 del programa de conservación de tortugas marinas en Laguna de Términos, Campeche, México. Contiene información de: 1. CPCTM Isla Aguada, 2. CPCTM Chacahito, 3. CPCTM Xicalango-Victoria y una Reseña estatal.

Guzmán-Hernández, V., García Alvarado, P.A., 2010. Informe téchnico 2009 del programa de conservación de tortugas marinas en Laguna de Términos, Campeche, México. Reporte especial para el APFFLT con información de: 1. CPCTM Xicalango-Victoria 2. CPCTM Chacahito, 3. CPCTM Isla Aguada, y 4. Reseña estatal r.

Guzmán-Hernández, V., García Alvarado, P.A., 2011. Informe técnico 2010 del programa de conservación de tortugas marinas en Laguna de Términos, Campeche, Mexico. Reporte especial para el APFFLT con información de: 1. CPCTM Xicalango-Victoria 2. CPCTM Chacahito, 3. CPCTM Isla Aguada, y 4. Reseña estatal re.

Guzmán-Hernández, V., García Alvarado, P.A., 2013a. Informe Técnico 2011 del programa de Conservación de Tortugas Marinas en Laguna de Términos, Campeche, México. Contiene información de: 1. CPCTM Xicalango-Victoria, 2. CPCTM Chacahito, 3. CPCTM Isla Aguada y 4. Reseña estatal regional. APFFLT/RPCyGM/CONANP.

Guzmán-Hernández, V., García Alvarado, P.A., 2013b. Informe Técnico 2012 del programa de conservación de tortugas marinas en Laguna de Términos, Campeche, México. Contiene información de: 1. CPCTM Xicalango-Victoria, 2. CPCTM Chacahito, 3. CPCTM Isla Aguada y 4. Reseña estatal regional. APFFLT/RPCyGM/CONANP.

Guzmán-Hernández, V., Nolasco Soto, J., Garcia Alvarado, P., 2008. Informe técnico final 2007 del programa conservación de tortugas marinas en Laguna de Terminos, Campeche, Mexico. Contiene información de los centros para la conservación de las tortugas marinas en Isla Aguada, INE/DGVS/TM-007-CAMP, Xicalango-Victoria y u.

Guzmán-Hernández, V., Rejón Puch, J.C., Gómez Gómez, R., Silva Sánchez, J., 1994. Informe final del programa de investigación y protección de las tortugas marinas del estado de campeche, México., temporada 1993. Panorama regional., Inédito.

Guzmán-Hernández, V., Rejón Puch, J.C., Gómez Gómez, R., Silva Sánchez, J., 1995a. Informe final del programa de investigación y protección de las tortugas marinas del estado de Campeche, México. Temporada 1994. Situación actual.

Guzmán-Hernández, V., Rejón Puch, J.C., Gómez Gómez, R., Silva Sánchez, J., Castro Castro, L., 1995b. Resumen del programa de investigación y protección de las tortugas marinas realizado en Campeche, Mexico, durante la temprada de andicación de 1995.

Guzmán-Hernández, V., Sanchez Perez, J.M., Gómez Gómez, R., Rejón Puch, J.C., Silva Sánchez, J., 1993. Informe final del programa tortugas marinas de Isla Aguada Carmen, Campeche. Temporada 1992. Una perspectiva regional., Inédito.

Hachiglou, V., Kolinski, S.P., Smith, A., 1991. Outer Island Turtle Project, Yap State, F.S.M., Status Report, in: The South Pacific Regional Environment Programme's Regional Marine Turtle Conservation Programme Meeting, August 1991, Noumea, New Caledonia.

Halim, M.H., Dermawan, A., 1999. Marine turtle research, management and conservation in Indonesia, in: Report of SEAFDEC-ASEAN Regional Workshop on Sea Turtle Conservation and Management. Kuala Trengganu, Malaysia, p. 328 pp.

Halim, M.H., Silalahi, S., Sugarjito, J., 2001. Conservation and utilization trend of marine turtles in Indonesia. Tigerpaper (FAO) 28, 10–18.

Hamabata, T., Nishida, S., Kamezaki, N.I., Koike, H., 2009. Genetic structure of populations of the green turtle (*Chelonia mydas*) in Japan using mtDNA control region sequences. Bull. Grad. Sch. Soc. Cult. Stud. Kyushu Univ. 15, 35–50.

Hamann, M., Cuong, C.T., Hong, N.D., Thuoc, P., Thuhien, B.T., 2006a. Distribution and abundance of marine turtles in the Socialist Republic of Viet Nam. Biodivers. Conserv. 15, 3703–3720.

Hamann, M., Hien, B.T.T., Cox, N., Thompson, J., Schäuble, C., Cuong, C.T., Khuong, T.C., Hong, N.D., 2005. Marine Turtle Conservation in Viet Nam – Towards 2010. Mar. Turt. Newsl. 5–6.

Hamann, M., Limpus, C.J., Hughes, G.R., Mortimer, J.A., Pilcher, N.J., 2006b. Assessment of the conservation status of the leatherback turtle in the Indian Ocean and South-East Asia, IOSEA Marine Turtle MoU Secretariat, Bangkok. IOSEA Marine turtle MoU Secretariat, Bangkok, Thailand.

Hamza, A., 2010. Libya, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Priorities. IUCN, Gland, Switzerland, pp. 157–170.

Hanafy, M.H., 2012. Nesting of marine turtles on the Egyptian beaches of the Red Sea. Egypt. J. Acquat. Biol. Fish. 16, 59–71.

Hanski, I., Gilpin, M., 1991. Metapopulation dynamics: brief history and conceptual domain. Biol. J. Linn. Soc. 42, 3–16.

Hare, S.R., 1991. Turtles caught incidental to demersal finfish fishery in Oman. Mar. Turt. Newsl. 53, 14–16.

Hariri, K.I., Nichols, P., Krupp, F., Mishrigi, S., Barrania, A., Ali, A.F., Kedidi, S.M., 2000. Status of the living marine resources in the Red Sea and Gulf of Aden region and their management.

Harley, C.D.G., Randall Hughes, a, Hultgren, K.M., Miner, B.G., Sorte, C.J.B., Thornber, C.S., Rodriguez, L.F., Tomanek, L., Williams, S.L., 2006. The impacts of climate change in coastal marine systems. Ecol. Lett. 9, 228–41.

Harrison, E., Troëng, S., 2005. Report on the 2004 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation and the Ministry of Envirionment, Engery and Telecommunications of Costa rica.

Hart, K.M., Zawada, D.G., Fujisaki, I., Lidz, B.H., 2013. Habitat use of breeding green turtles *Chelonia mydas* tagged in Dry Tortugas National Park: Making use of local and regional MPAs. Biol. Conserv. 161, 142–154.

Harvey, T., Townsend, S., Kenyon, N., Redfern, G., 2005. Monitoring of nesting sea turtles in the Coringa Herald National Nature Reserve: 1991/92–2003/04 Nesting Seasons. Australia: Indo-Pacific Sea Turtle Conservation Group. Final report.

Hasan, M.M., 2009. Tourism and conservation of biodiversity: A case study of St. Martins Island, Bangladesh. Law, Soc. Justice Glob. Dev. J. 2009.

Hasbún, C.R., Lawrence, A.J., Samour, J.H., Al-Ghais, S.M., 2000. Preliminary observations on the biology of green turtles, *Chelonia mydas*, from the United Arab Emirates. Aquat. Conserv. Mar. Freshw. Ecosyst. 10, 311–322.

Hatase, H., Sato, K., Yamaguchi, M., Takahashi, K., Tsukamoto, K., 2006. Individual variation in feeding habitat use by adult female green sea turtles (*Chelonia mydas*): are they obligately neritic herbivores? Oecologia 149, 52–64.

Havea, S., MacKay, K.T., 2009. Marine Turtle Hunting in the Ha'apai Group, Tonga. Mar. Turt. Newsl. 123, 15–17.

Hawkes, L.A., Broderick, A.C., Godfrey, M.H., Godley, B.J., 2007. Investigating the potential impacts of climate change on a marine turtle population. Glob. Chang. Biol. 13, 923–932.

Hawkes, L.A., Broderick, A.C., Godfrey, M.H., Godley, B.J., 2009. Climate change and marine turtles. Endanger. Species Res. 7, 137–154.

Haxhiu, I., 2010. Albania, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Priorities. IUCN, Gland, Switzerland, pp. 15–28.

Haxhiu, I., Rumano, M., 2006a. Stavnik as a valuable fishing means for the protection of marine organisms, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Proceedings of the Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece, p. 247.

Haxhiu, I., Rumano, M., 2006b. *Chelonia mydas* (LINNAEUS, 1758) gjendet për herë të parë në bregdetin e Shqipërisë [*Chelonia mydas* (LINNAEUS, 1758) found for the first time in Albania coast]. Bul. i Shk. [Bulletin Sci. 56, 153–157.

Hay, E., Sablan-Zebedy, E., 2005. USE Republic of the Marshall Islands: country environmental analysis. ADB Technical Assistance Consultant's Report.

Haynes-Sutton, A., Bjorkland, R.K., Donaldson, A., 2011. WIDECAST Sea Turtle Recovery Action Plan for Jamaica. CEP Technical Report No. 50 UNEP Caribbean Environment Programme, Kingston, Jamaica.

Hays, G.C., Glen, F., Broderick, A.C., Godley, B.J., Metcalfe, J.D., 2002. Behavioural plasticity in a large marine herbivore: contrasting patterns of depth utilisation between two green turtle (*Chelonia mydas*) populations. Mar. Biol. 141, 985–990.

Helvey, M., Fahy, C.C., 2012. Fisheries management off the U.S. West Coast: a progressive model for sea turtle conservation, in: Seminoff, J.A., Wallace, B.P. (Eds.), Sea Turtles of the Eastern Pacific: Advances in Research and Conservation. University of Arizona Press, Tucson, Arizona, pp. 113–135.

Hendrickson, J.R., 1958. The green sea turtle, *Chelonia mydas* (Linn.) in Malay and Sarawak. Proc. Zool. Soc. London 130, 455–535.

Henwood, T.A., Ogren, L.H., 1987. Distributions and migrations of immature Kemp's ridley turtles (Lepidochelys kempi) and green turtles (*Chelonia mydas*) off Florida, Georgia, and South Carolina. Northeast Gulf Sci. 9, 153–159.

Henwood, T.A., Stuntz, W.E., 1987. Analysis of sea turtle captures and mortalities during commercial shrimp trawling. Fish. Bull. 85, 813–817.

Herbst, L.H., 1994. Fibropapillomatosis of marine turtles. Annu. Rev. Fish Dis. 4, 389–425.

Herbst, L.H., Klein, P.A., 1995. Green turtle fibropapillomatosis: Challenges to assessing the role of environmental cofactors, in: Rolland, R., Gilbertson, M., Colborn, T. (Eds.), Wildlife Development. Work Session on Environmentally Induced Alterations in Development: A Focus on Wildlife (Racine, Wisconsin; 10-12 December 1993). Environmental Health Perspectives, Vol. 103, Supplement 4. pp. 27–30.

Hermanussen, S., Matthews, V., Päpke, O., Limpus, C.J., Gaus, C., 2008. Flame retardants (PBDEs) in marine turtles, dugongs and seafood from Queensland, Australia. Mar. Pollut. Bull. 57, 409–418.

Herren, R.M., 1999. The effect of beach nourishment on loggerhead (*Caretta caretta*) nesting and reproductive success at Sebastian Inlet, Florida. University of Central Florida, Orlando, Florida.

Hien, T.M., 2002. Status of sea turtle conservation in Vietnam, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. February 5–8, 2002. Honolulu, HI: Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 191–194.

Hildebrand, H., 1987. A reconnaissance of beaches and coastal waters from the border of Belize to the Mississippi River as habitats for marine turtles. SEFC Panama City Laboratory.

Hildebrand, H.H., 1982. A historical review of the sea turtle populations in the Western Gulf of Mexico, in: Bjorndal, K. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C., pp. 447–453.

Hilterman, M.L., Goverse, E., de Bres, J., 2001. The sea turtles of Suriname, 2000, WWF. Biotopic in collaboration with the Foundation for Nature Conservation Suriname (STINASU).

Hirama, S., 2001. Epizootiology of fibropapillomatosis in green turtle on the Atlantic coast of Florida. University of Central Florida, Orlando, Florida.

Hirama, S., Ehrhart, L.M., 2007. Description, prevalence and severity of green turtle fibropapillomatosis in three developmental habitats on the east coast of Florida. Florida Sci. 70, 435–448.

Hirth, H.F., 1968. The green turtle resource of South Arabia, and the status of the green turtle in the Seychelles Islands, Yemen and the Seychelles Islands on the green turtle. Report to the governments of the People's Republic of Southern Yemen, and the Seychelles Islands. FAO, Rep. FAO/UNDP(TA), (2467). 58 pages.

Hirth, H.F., 1980. Some Aspects of the Nesting Behavior and Reproductive Biology of Sea Turtles. Integr. Comp. Biol. 20, 507–523.

Hirth, H.F., 1997. Synopsis of the biological data on the green turtle *Chelonia mydas* (Linnaeus 1758). Fish and Wildlife Service, Washington, D.C, Biological Report 97(1), 120 pages.

Hirth, H.F., Carr, A.F., 1970. The green turtle in the Gulf of Aden and Seychelles Islands. Verh. der K. Ned. Akad. van Wetnshchappen. Afd. Natuurkd. 58, 1–44.

Hirth, H.F., Hollingsworth, S.L., 1973. Report to the Government of the People's Democratic Republic of Yemen on marine turtle management. United Nations Development Programme No. TA 3178.

Hirth, H.F., Rohovit, D.L.H., 1992. Marketing patterns of green and hawksbill turtles in Port Moresby, Papua New Guinea. Oryx 26, 39–42.

Hitipeuw, C., Maturbongs, J., 2002. Marine turtle conservation program Jamurba-Medi nesting beach, north coast of the Bird's Head Peninsula, Papua, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. February 5–8, 2002. Honolulu, HI: Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 161–175.

Hitipeuw, C., Mustofa, I., Soede, L.P., 2006. Preliminary assessment of sea turtle interactions with Indonesian fisheres, in: Book of Abstracts of the Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation2. International Sea Turtle Society, Athens, Greece, pp. 249–249.

Hitipeuw, C., Pet-Soede, L., 2004. A need to align and integrate incentive strategies, in: Expert Consultation on Interactions between Sea Turtles and Fisheries within an Ecosystem Context. FAO Fisheries Report. No. 738, Supplement, FAO, Rome, p. 238 pp.

Hochscheid, S., Bentivegna, F., Bradai, M.N., Hays, G.C., 2007. Overwintering behaviour in sea turtles: dormancy is optional. Mar. Ecol. Prog. Ser. 340, 287–298.

Holroyd, G.L., Trefry, H.E., 2010. The Importance of Isla Clarión, Archipelago Revillagigedo, Mexico, for Green Turtle (*Chelonia mydas*) Nesting. Chelonian Conserv. Biol. 9, 305–309.

Hopkins, S.R., Murphy, T.M., 1980. Reproductive ecology of *Caretta caretta* in South Carolina. South Carolina Wildl. Mar. Resour. Dep. Complet. Rep.

Horikoshi, K., Suganuma, H., Tachikawa, H., Sato, F., Yamaguchi, M., 1994. Decline of Ogasawara green turtle population in Japan, in: Bjorndal, K.A., Bolten, A.B., Johnson, D.A., Eliazar, P.J. (Eds.), Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. US Department of Commerce, NOAA Technical Memorandum NMFS-SEFSC-351, Miami, FL, pp. 235–236.

Hosier, P.E., Kochhar, M., Thayer, V., 1981. Off-road vehicle and pedestrian track effects on the sea-approach of hatchling loggerhead turtles. Environ. Conserv. 8, 158–161.

Hossain, M.G., Hoq, M.E., 2010. Sustainable management of fisheries resources of the Bay of Bengal-Compilation of national and regional workshop Reports. Support to Sustain. Manag. BOBLME Proj. Bangladesh Fish. Res. Institute. SBOBLMEP Pub./Rep.2. 122 p.

Howe, S.A., Asfaha, B., Kemp, J.M., 2004. Turtle Strandings in the southern Eritrean Red Sea. Mar. Turt. Newsl. 103, 4–7.

Howell, K.M., Mbindo, C., 1996. The status of sea turtle conservation in Tanzania, in: Humphrey, S.L., Salm, R.V. (Eds.), Status of Sea Turtle Conservation in the Western Indian Ocean. UNEP Regional Seas Reports and Studies No. 165, Nairobi, Kenya, p. 162.

Howell, L.N., Reich, K.J., Shaver, D.J., Landry, A.M., 2013. Integration of gut content and stable isotope analysis to investigate ontogenetic shifts in diet and habitat by juvenile green sea turtles (*Chelonia mydas*) along the Texas coast, in: Tucker, T., Belskis, L., Panagopoulou, A., Rees, A., Frick, M., Williams, K., LeRoux, R., Stewart, K. (Eds.), Proceedings of the Thirty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NOAA-SEFSC-645, pp. 141–142.

Hughes, A.L., Caine, E.A., 1994. The effects of beach features on hatcling loggerhead sea turtles, in: Bjorndal, K.A., Bolten, A.B., Johnson, D.A., Eliazar, P.J. (Eds.), Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-351, p. 237.

Hughes, G., 1970. The status of sea turtles in South East Africa, Madagascar and the Mascarenes. Eur. Island. Roneo. Rept, Ocenogr. Inst. Durban 1–47.

Hughes, G.R., 1974. The sea turtles of South East Africa II. Investigational report 36. South African Association of Marine Biological Research, Durban.

Hughes, G.R., 1981. Conservation of Sea Turtles in the Southern Africa Region, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press Book, Washington D.C., pp. 397–404.

Humber, F., Godley, B.J., Ramahery, V., Broderick, A.C., 2010. Using community members to assess artisanal fisheries: the marine turtle fishery in Madagascar. Anim. Conserv. 2010, 1–11.

Humphrey, S.L., Salm, R.V., 1996. Status of Sea Turtle Conservation in the Western Indian Ocean, in: UNEP Regional Seas Report and Studies No. 165. IUCN/UNEP. Nairobi, Kenya, p. 162.

Hunter, M.L., Gibbs, J.P., 2007. Fundamentals of Conservation Biology, Third Edit. ed. Blackwell Publishing.

Hykle, D., 2002. The Convention on Migratory Species and other international instruments relevant to marine turtle conservation: pros and cons. J. Int. Wildl. Law Policy 5, 105–119.

Hykle, D., 2012. Sea Turtle Conservation Center of the Royal Thai Navy. Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding.

Hykle, D., 2013. IOTC / IOSEA reports give insights into Indian Ocean fisheries-turtle interactions [WWW Document]. Indian Ocean - South-East Asian Mar. Turt. Memo. Underst. Profile Mon. URL http://www.ioseaturtles.org/pom_detail.php?id=127

Ikonomopoulou, M.P., Olszowy, H., Francis, R., Ibrahim, K., Whittier, J.M., 2013. Accumulation of trace metals in the embryos and hatchlings of *Chelonia mydas* from Peninsular Malaysia incubated at different temperatures. Sci. Total Environ. 450-451, 301–306.

Ikonomopoulou, M.P., Olszowy, H., Hodge, M., Bradley, A.J., 2009. The effect of organochlorines and heavy metals on sex steroid-binding proteins in vitro in the plasma of nesting green turtles, *Chelonia mydas*. J. Comp. Physiol. B Biochem. Syst. Environ. Physiol. 179, 653–662.

Innocenzi, J., Maury, J., M'soili, A., Ciccione, S., 2010. Reproduction biology of green turtle in Itsamia, Mohéli (Union of Comoros). Indian Ocean Turt. Newsl. 11, 5–7.

IPCC, 2007. Summary for Policymakers, in: Solomon, S., Quin, D., Manning, M., Chen, Z., Marquis, M., Averyt, K.B., Tignor, M., Miller, H.L. (Eds.), Climate Change 2007: The Physical Science Basis. Contribution of Working Group I to the Fourth Assessment Report of the Intergovernmental Panel on Climate Change. Cambridge Univ Press, Cambridge, United Kingdom and New York, New York, USA.

Isamu, T., Guilbeaux, M.D., 2002. USE Sea turtle conservation in Palau, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, pp. 195–196.

Ishizaki, A., 2007. Toward reducing human-caused impacts on green turtle (*Chelonia mydas*) nesting activity in Ogasawara Islands, Japan. Colorado State University, Fort Collins, Colorado.

Ishizaki, A., Teel, T.L., Yamaguchi, M., 2011. Contextual factors influencing support for sea turtle management actions in Ogasawara Islands, Japan: an application of conjoint analysis. Hum. Dimens. Wildl. 16, 287–298.

Islam, M.Z., 2001. Notes on the trade in marine turtle products in Bangladesh. Mar. Turt. Newsl. 94, 10.

Islam, M.Z., 2002. Marine turtle nesting at St. Martin's Island, Bangladesh. Mar. Turt. Newsl. 96, 19–21.

Islam, M.Z., Ehsan, F., Rahman, M.M., 2011. Nesting sea turtles at Sonadia Island, Bangladesh. Mar. Turt. Newsl. 130, 19–22.

Jacob, P., 2000. The status of marine resources and coral reefs of Nauru.

Japan Fisheries Resource Conservation Association, 1999. Research on migratory routes and diving ecology of sea turtles. General report on commissioned project for conservation research of aquatic animals in 1994–1998, Vol 1. Japan Fisheries Resource Conservation Association, Tokyo, p 1–90 (in Japanese).

Jensen, M.P., 2010. Assessing the composition of green turtle (*Chelonia mydas*) foraging grounds in Australasia using mixed stock analyses. Ph.D. Thesis, Institute for Applied Ecology, Univesity of Canberra.

Jensen, M.P., FitzSimmons, N.N., Dutton, P.H., 2013. Molecular genetics of sea turtles, in: Wyneken, J., Lohmann, K.J., Musick, J.A. (Eds.), The Biology of Sea Turtles, Volume 3. CRC Press, Boca Raton, FL, pp. 135–154.

Jit, J.N., 2007. Status of sea turtle conservation in Fiji: an assessment of the international, regional and national Focus. School of Marine Studies, University of the South Pacific.

Johnson, S.A., Ehrhart, L.M., 1995. Reproductive biology of the Florida green turtle (*Chelonia mydas*) at Melbourne Beach, Florida, 1991, in: Richardson, J.I., Richardson, T.H. (Eds.), Twelfth Annual Workshop on Sea Turtle Biology and Conservation. U.S. Department of Commerce, NOAA Technical Memorandum NMFS, NMFS-SEFSC-361, pp. 54–56.

Johnson, S.A., Ehrhart, L.M., 1996. Reproductive ecology of the Florida green turtle: clutch frequency. J. Herpetol. 30, 407–410.

Jones, S., Fernando, A.B., 1973. The present status of the turtle fishery in the Gulf of Mannar and Palk Bay, in: Proceedings of the Symposium of Living Resources of the Seas around India, 1968, Mandapam Camp. CMFRI Special Publication, pp. 712–715.

Jones, T.T., Seminoff, J.A., 2013. Feeding biology: advances from field-based observations, physiological studies and molecular techniques, in: Wyneken, J., Lohmann, K.J., Musick, J.A. (Eds.), The Biology of Sea Turtles Volume III. CRC Press, Boca Raton, FL, pp. 211–247.

Jony, M., Rees, A.F., 2008. Three years of monitoring turtle nesting in Syria (2004-2006): what progress has been made toward protection, in: Rees, A.F., Frick, M., Panagopoulou, A., Williams, K. (Eds.), Proceedings of the Twenty-Seventh Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-569, pp. 238–239.

Juarez, R., Muccio, C., 1997. Sea turtle conservation in Guatemala. Mar. Turt. Newsl. 77, 15–17.

Juarez-Ceron, J.A., Sarti-Martinez, A.L., Dutton, P.H., 2003. First study of the green/black turtles of the Revillagigedo Archipelago: a unique nesting stock in the Eastern Pacific, in: Seminoff, J.A. (Ed.), Twenty- Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL, p. 70.

Juvik, S., Juvik, J., Thomas, R., 1999. Atlas of Hawaii. University Press of Hawaii.

Kabraji, A.M., Firdous, F., 1984. Conservation of turtles, Hawkesbay and Sandspit, Pakistan.

Kabua, E., Langidrik, M., Boktok, K., Parker, D.M., Frutchey, K.P., 2012. Marine turtle-fisheries interaction mitigation training, data collection activities and satellite telemetry in the Republic of the Marshall Islands, in: Belskis, L., Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Proceedings of the Twenty-Ninth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NOAA NMFS-SEFSC-630, Brisbane, Australia, p. 120.

Kamezaki, N.I., Matsui, M., 1995. Geographic variation in skull morphology of the green turtle, *Chelonia mydas*, with a taxonomic discussion. J. Herpetol. 29, 51–60.

Kamezaki, N.I., Matsuzawa, Y., Abe, O., Asakawa, H., Fujii, T., Goto, K., Hagino, S., Hayami, M., Ishii, M., Iwamoto, T., Kamata, T., Kato, H., Kodama, J., Kondo, Y., Miyawaki, I., Mizobuchi, K., Nakamura, Y., Nakashima, Y., Naruse, H., Omuta, K., Samejima, M., Suganuma, H., Takeshita, H., Tanaka, T., Toji, T., Uematsu, M., Yamamoto, A., Yamato, T., Wakabayashi, I., 2003. Loggerhead turtles nesting in Japan, in: Bolten, A.B., Witherington, B.E. (Eds.), Loggerhead Sea Turtles. Smithsonian Institution Press, Washington, D.C, pp. 210–217.

Kamrowski, R.L., Limpus, C.J., Jones, R., Anderson, S., Hamann, M., 2014. Temporal changes in artificial light exposure of marine turtle nesting areas. Glob. Chang. Biol.

Kaneko, J.J., Bartram, P.K., 2008. What If You Don’t Speak “CPUE-ese”? Pelagic Fish. Res. Progr. 13, 8 pages.

Kapurusinghe, T., 2006. Status and conservation of marine turtles in Sri Lanka, in: Shanker, K., Choudhury, B.C. (Eds.), Marine Turtles of the Indian Subcontinent. Universities Press, Hyderabad, India, pp. 173–187.

Kapurusinghe, T., Cooray, R., 2002. Marine turtle by-catch in Sri Lanka: survey report September 1999-November 2000.

Kapurusinghe, T., Ekanayake, E.M.L., Saman, M.M., Rathnakumara, D.S., 2005. Community-based marine turtle conservation in Kosgoda, Sri Lanka: nesting results from 2005 to 2007. Testudo 6.

Kar, C.S., Bhaskar, S., 1982. Status of sea turtles in the Eastern Indian Ocean, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C., pp. 365–372.

Karl, S.A., Bowen, B.W., Avise, J.C., 1992. Global population genetic structure and male-mediated gene flow in the green turtle (*Chelonia mydas*): RFLP analyses of anonymous nuclear loci. Genetics 131, 163–173.

Kaska, Y., 2008. Meeting report: Second Turkish National Sea Turtle Symposium (25-27 October 2007). Mar. Turt. Newsl. 20–22.

Kaska, Y., Ilgaz, C., Ozdemir, A., Başkale, E., Türkozan, O., Baran, I., Stachowitsch, M., 2006. Sex ratio estimations of loggerhead sea turtle hatchlings by histological examination and nest temperatures at Fethiye beach, Turkey. Naturwissenschaften 93, 338–43.

Kasparek, M., 2007. Green Turtle (*Chelonia mydas*) Conservation Monitoring in Kazanli (Turkey).

Kasparek, M., Godley, B.J., Broderick, A.C., 2001. Nesting of the Green Turtle, *Chelonia mydas*, in the Mediterranean: a review of status and conservation needs. Zool. Middle East 24, 45–74.

Kelez, S., Velez-Zuazo, X., Manrique, C., 2003. New evidence on the Loggerhead sea turtle *Caretta caretta* (Linnaeus 1758) in Peru. Ecol. Apl. 2, 141–142.

Kelkar, N., Arthur, R., Marba, N., Alcoverro, T., 2013. Green turtle herbivory dominates the fate of seagrass primary production in the Lakshadweep islands (Indian Ocean). Mar. Ecol. Prog. Ser. 485, 235–243.

Kelkar, N., Arthur, R., Marbà, N., Alcoverro, T., 2013. Greener pastures? High-density feeding aggregations of green turtles precipitate species shifts in seagrass meadows. J. Ecol. 101, 1158–1168.

Keller, B.D., Gleason, D.F., McLeod, E., Woodley, C.M., Airame, S., Causey, B.D., Friedlander, A.M., Grober-Dunsmore, R., Johnson, J.E., Miller, S.L., Steneck, R.S., 2009. Climate Change, Coral Reef Ecosystems, and Management Options for Marine Protected Areas. Environ. Manage. 44, 1069–1088.

Kelliher, S., Lusty, C., Prophet, M., 2011. Ningaloo turtle program annual report 2010-2011. Ningaloo Turtle Program, Exmouth, Western Australia.

Kendall, B.E., 2009. The diffusion approximation overestimates the extinction risk for count-based PVA. Conserv. Lett. 2, 216–225.

Kennett, R., N., M., Yunupingu, D., 1998. Migration patterns of marine turtles in the Gulf of Carpentaria, Northern Australia: Implications for Aboriginal management. Wildl. Res. 31, 241–248.

Khalil, M., Syed, H., Aureggi, M., Venizelos, L., 2009. Marine turtle nesting at El Mansouri, South Lebabanon, in: Demetropoulos, A., Turkozan, O. (Eds.), Proceedings of the Second Mediterranean Conference on the Sea Turtles. Barcelona Convention – Bern Convention – Bonn Convention (CMS), Kemer, Turkey, pp. 109–112.

Khan, A., Nishat, A., Ahmed, R., 2006. Conservation assessment and management action plan for marine turtles in Bangladesh, in: Pilcher, N.J. (Ed.), Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-536, p. 60.

Khan, M.A.R., 1982. Wildlife of Bangladesh. Univ. Dhaka, Dhaka.

Kikukawa, A., Kamezaki, N.I., 1999. Current status of the sea turtles nesting on Okinawajima and adjacent islands of the central Ryukyus, Japan. Biol. Conserv. 87, 149–153.

Kinch, J., 2006. Socioeconomic baseline study of communities involved in leatherback turtle nesting beach projects along the Huon Coast, Morobe Province, Papua New Guinea. Honolulu, Hawaii, USA.

King, F.W., 1982. Historical review of the decline of the green turtle and the hawksbill, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C., pp. 183–188.

King, R., Cheng, W., Tseng, C.-T., Chen, H., Cheng, I.-J., 2013. Estimating the sex ratio of green sea turtles (*Chelonia mydas*) in Taiwan by the nest temperature and histological methods. J. Exp. Mar. Bio. Ecol. 445, 140–147.

Kiribati Ministry of Environment Lands and Agriculture Development, 2009. Republic of Kiribati national capacity self assessment project: assessing the capacity of the Republic of Kiribati to implement the United Nations Convention on Biological Diversity (UNCBD), United Nations Convention to Combat Desertification (UNCCD) and. the United Nations Framework Convention on Climate Change (UNFCCC). Final Report, November 2009. 61 pages.

Kittinger, J.N., Pandolfi, J.M., Blodgett, J.H., Hunt, T.L., Jiang, H., Maly, K., McClenachan, L.E., Schultz, J.K., Wilcox, B.A., 2011. Historical reconstruction reveals recovery in Hawaiian coral reefs. PLoS One 6, e25460.

Kittinger, J.N., Van Houtan, K.S., McClenachan, L.E., Lawrence, A.L., 2013. Using historical data to assess the biogeography of population recovery. Ecography (Cop.). 36, 868–872.

Klain, S., Eberdong, J., Kitalong, A., Yalap, Y., Matthews, E., Eledui, A., Morris, M., Andrew, W., Albis, D., Kemesong, P., 2007. Linking Micronesia and Southeast Asia: Palau sea turtle satellite tracking and flipper tag returns. Mar. Turt. Newsl. 118, 9–11.

Kobayashi, D.R., Polovina, J.J., Parker, D.M., Kamezaki, N.I., Cheng, I.-J., Uchida, I., Dutton, P.H., Balazs, G.H., 2008. Pelagic habitat characterization of loggerhead sea turtles, *Caretta caretta*, in the North Pacific Ocean (1997-2006): Insights from satellite tag tracking and remotely sensed data. J. Exp. Mar. Bio. Ecol. 356, 96–114.

Koch, V., Brooks, L.B., Nichols, W.J., 2007. Population ecology of the green/black turtle (*Chelonia mydas*) in Bahía Magdalena, Mexico. Mar. Biol. 153, 35–46.

Koch, V., Nichols, W.J., Peckham, S.H., Delatoba, V., 2006. Estimates of sea turtle mortality from poaching and bycatch in Bahía Magdalena, Baja California Sur, Mexico. Biol. Conserv. 128, 327–334.

Kolinski, S.P., 1994. Carapace lesions of *Chelonia mydas* breeding in Yap State are diagnosed to be fibropapilloma. Mar. Turt. Newsl. 67, 26–31.

Kolinski, S.P., Hoeke, R.K., Holzwarth, S.R., Ilo, L.I., Cox, E.F., O'Conner, R.C., Vroom, P.S., 2006. Nearshore distribution and an abundance estimate for green sea turtles, *Chelonia mydas*, at Rota Island, Commonwealth of the Northern Mariana Islands. Pacific Sci. 60, 509–522.

Kolinski, S.P., Hoeke, R.K., Holzwarth, S.R., Vroom, P.S., 2005. Sea Turtle Abundance at Isolated Reefs of the Mariana Archipelago. Micronesica 37, 287–296.

Kolinski, S.P., Ilo, L.I., Manglona, J.M., 2004. Green Turtles and Their Marine Habitats at Tinian and Aguijan, with Projections on Resident Turtle Demographics in the Southern Arc of the Commonwealth of the Northern Mariana Islands. Micronesica 37, 97–118.

Kolinski, S.P., Parker, D.M., Ilo, L.I., Ruak, J.K., 2001. An Assessment of the Sea Turtles and Their Marine and Terrestrial Habitats at Saipan, Commonwealth of the Northern Mariana Islands. Micronesica 34, 55–72.

Komar, P.D., 1983. Coastal erosion in response to the construction of jetties and breakwaters, in: Komar, P.D. (Ed.), CRC Handbook of Coastal Processes and Erosion. CRC Press, Boca Raton, Florida, pp. 191–204.

Komoroske, L.M., Lewison, R.L., Seminoff, J.A., Deheyn, D.D., Dutton, P.H., 2011. Pollutants and the health of green sea turtles resident to an urbanized estuary in San Diego, CA. Chemosphere 84, 544–52.

Komoroske, L.M., Lewison, R.L., Seminoff, J.A., Deustchman, D.D., Deheyn, D.D., 2012. Trace metals in an urbanized estuarine sea turtle food web in San Diego Bay, CA. Sci. Total Environ. 417-418, 108–16.

Kopsida, H., Margaritoulis, D., Dimopoulos, D., 2002. What marine turtle strandings can tell us, in: Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-477, pp. 207–209.

Kowarsky, J., 1982. Subsistence hunting of sea turtles in Australia, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C., pp. 305–313.

Krahn, M.M., Wade, P.R., Kalinowski, S.T., Dahlheim, M.E., Taylor, B.L., Hanson, M.B., Ylitalo, G.M., Angliss, R.P., Stein, J.E.E., Waples, R.S.S., Ford, M.J., Perrin, W.F., 2004. 2004 Status review of Southern Resident killer whales (Orcinus orca) under the endangered species act. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-62 73.

Krogh, M., Reid, D., 1996. Bycatch in the protective shark meshing programme off south-eastern New South Wales, Australia. Biol. Conserv. 77, 219–226.

Kubis, S.A., Chaloupka, M.Y., Ehrhart, L.M., Bresette, M.J., 2009. Growth rates of juvenile green turtles *Chelonia mydas* from three ecologically distinct foraging habitats along the east central coast of Florida, USA. Mar. Ecol. Prog. Ser. 389, 257–269.

Kumoru, L., 2008. Tuna fisheries report – Papua New Guinea.

Kuroyanagi, K., Kamezaki, N.I., Sugiyama, S., 1999. First confirmed black sea turtle (C. m. agassizii) in Japanese waters. Umigame Newsl. Japan 39, 13–14.

Lagueux, C.J., 1998. Marine turtle fishery of Caribbean Nicaragua: Human use patterns and harvest trends. University of Florida, Gainesville.

Lagueux, C.J., Campbell, C.L., 2005. Marine turtle nesting and conservation needs on the south-east coast of Nicaragua. Oryx 39, 398.

Lagueux, C.J., Campbell, C.L., Coulson, E., Coulson, A., 2012. Conservation and monitoring of sea turtle nesting on the southeast cost of Nicaragua, 2008-2011. Final Report. Wildl. Conserv. Soc. Pearl Lagoon, RAAS, Nicar. 19 pages.

Lahanas, P.N., Bjorndal, K.A., Bolten, A.B., Encalada, S.E., Miyamoto, M.M., Valverde, R.A., Bowen, B.W., 1998. Genetic composition of a green turtle (*Chelonia mydas*) feeding ground population: evidence for multiple origins. Mar. Biol. 130, 345–352.

Lahanas, P.N., Miyamoto, M.M., Bjorndal, K.A., Bolten, A.B., 1994. Molecular evolution and population genetics of Greater Caribbean green turtles (*Chelonia mydas*) as inferred from mitochondrial DNA control region sequences. Genetica 94, 57–66.

Lal, A., Arthur, R., Marbà, N., Lill, A.W.T., Alcoverro, T., 2010. Implications of conserving an ecosystem modifier: Increasing green turtle (*Chelonia mydas*) densities substantially alters seagrass meadows. Biol. Conserv. 143, 2730–2738.

Lam, T., Ling, X., Takahashi, S., Burgess, E.A., 2011. Market Forces: An Examination of Marine Turtle Trade in China and Japan. TRAFFIC East Asia, Hong Kong.

Lapointe, B.E., 1999. Simultaneous and bottom-up forces control blooms on coral reefs macroalgal. Limnol. Oceanogr. 44, 1586–1592.

Laurent, L., Abd El-Mawla, E.M., Bradai, M.N., Demirayak, F., Oruç, A., 1996. Reducing sea turtle mortality induced by Mediterranean fisheries: trawling activity in Egypt, Tunisia, and Turkey. Rome, Italy.

Laurent, L., Camiñas, J.A., Casale, P., Deflorio, M., De Metrio, G., Kapantagakis, A., Margaritoulis, D., Politou, C.-Y., Valeiras, J., 2001. Assessing marine turtle bycatch in European drifting longline and trawl fisheries for identifying fishing regulations. Villeurbanne, France.

Laurent, L., Nouira, S., Jeudy De Grissac, A., Bradai, M.N., 1990. Les tortues marines de Tunisie: premières données. Bull. Soc. Herpetol. Fr. 53, 1–17.

Laurent, L., Olivier, G., Nougarede, J.P., Groul, J.-M., Robert, P., Cheylan, M., Finelli, F., Bompar, J.M., Dhermain, F., 1997. Observations de tortues marines en Méditerranéeb Francaise: Données anciennes inédites, années 1996 et 1997. Faune de Provence (C.E.E.P.) 18, 95–101.

Lauret-Stepler, M., Bourjea, J., Roos, D., Pelletier, D., Ryan, P.G., Ciccione, S., Grizel, H., 2007. Reproductive seasonality and trend of *Chelonia mydas* in the SW Indian Ocean: a 20 yr study based on track counts. Endanger. Species Res. 3, 217–227.

Laveti, M., MacKay, K.T., 2009. Does Fiji's Turtle Moratorium Work? Mar. Turt. Newsl. 123, 12–15.

Lazar, B., 2010. Croatia, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Priorities. IUCN, Gland, Switzerland, pp. 29–40.

Lazar, B., Casale, P., Tvrtkovic, N., Kozul, V., Tutman, P., Glavic, N., 2004. The presence of the green sea turtle, *Chelonia mydas*, in the Adriatic Sea. Herpetol. J. 14, 143–147.

Le Gall, J.Y., Bosc, P., Chateau, D., Taquet, M., 1986. Estimation du nombre de tortues vertes femelles adultes *Chelonia mydas* par saison de ponte à Tromelin et Europa (Océan Indien) (1973-1985). Oceanogr. Trop. 21, 3–22.

Lebeau, A., 1985. Breeding evaluation trials in the Green Turtle *Chelonia mydas* (Linné) on Scilly Atoll (leeward Islands, French Polynesia) during the breeding seasons 1982-1983 and 1983-1984, in: Harmelin, V.M., Salvat, B. (Eds.), Proceedings of The Fifth International Coral Reef Congress. National Museum of Natural History and Practical School of Advanced Studies, Tahiti, French Polynesia, pp. 487–494.

Lebeau, A., Biais, G., Durand, J.L., Gobert, B., 1983. La tortue verte *Chelonia mydas* (Linne) des Isles de Tromelin et d'Europa (Ocean Indien): peuplement et reproduction. Inst. Scient. Techn. Pêches Marit., L Port Réunion. 39 pp.

Lee Long, W.J., Coles, R.G., McKenzie, L.J., 2000. Issues for seagrass conservation management in Queensland. Pacific Conserv. Biol. 5, 321–328.

Leh, C., 1994. Hatch rates of green turtle eggs in Sarawak. Hydrobiologia 285, 171–175.

Leigh, E.G., O'Dea, A., Vermeij, G.J., 2014. Historical biogeography of the Isthmus of Panama. Biol. Rev. Camb. Philos. Soc. 89, 148–172.

Lemons, G.E., Lewison, R.L., Komoroske, L.M., Gaos, A.R., Lai, C.-T., Dutton, P.H., Eguchi, T., LeRoux, R.A., Seminoff, J.A., 2011. Trophic ecology of green sea turtles in a highly urbanized bay: Insights from stable isotopes and mixing models. J. Exp. Mar. Bio. Ecol. 405, 25–32.

Leong, T., Barclay, A., Howard, B., Waller, L., 2003. Assessing the impact of fishing pier construction on spatial patterns of sea turtle nesting in Palm Beach County, Florida, in: Seminoff, J.A. (Ed.), Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503, p. 309.

LeRoux, R.A., Balazs, G.H., Dutton, P.H., 2003. Genetic stock composition of foraging green turtles off the southern coast of Molokai, Hawaii, USA., in: J, S. (Ed.), Proceedings of the Twenty Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Tech Memo NMFS SEFSC-503. National Technical Information Service, Springfield, VA, pp. 251–252.

Lessa, W.A., 1984. Sea turtles and ritual: conservations in the Caroline Islands, in: Gunda, B. (Ed.), The Fishing Culture of the World: Studies in Ethnology, Cultural Ecology and Folklore. Volume II. Akadémiai Kiadó, Budapest, Hungary, pp. 1183–1201.

Levy, Y., 2010. Israel, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Policies. IUCN, Gland, Switzerland, pp. 111–133.

Lewis, C.B., 1940. The Cayman Islands and marine turtle, in: Grant, C. (Ed.), The Herpetology of the Cayman Islands. Bulletin of the Institue of Jamaica Science Series, Kingston. pp. 56–65.

Lewison, R.L., Crowder, L.B., Read, A.J., Freeman, S.A.A., 2004. Understanding impacts of fisheries bycatch on marine megafauna. Trends Ecol. Evol. 19, 598–604.

Lewison, R.L., Crowder, L.B., Wallace, B.P., Moore, J.E., Cox, T., Zydelis, R., McDonald, S., DiMatteo, A., Dunn, D.C., Kot, C.Y., Bjorkland, R., Kelez, S., Soykan, C., Stewart, K.R., Sims, M., Boustany, A., Read, a. J., Halpin, P., Nichols, W.J., Safina, C., 2014. Global patterns of marine mammal, seabird, and sea turtle bycatch reveal taxa-specific and cumulative megafauna hotspots. Proc. Natl. Acad. Sci. 111, 5271–5276.

Lezama, C., 2009. impacto de la pesqueria artesanal sobre la tortoga verde (*Chelonia mydas*) en las costas del Rio de la Plata exterior. Universidad de la República.

Liew, H.-C., 2002. Status of marine turtle conservation and research in Malaysia, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. February 5–8, 2002. Honolulu, HI: Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 51–56.

Lilette, V., 2006. Conservation et patrimonialisation de la tortue marine dand le sud ouest de l'ocean Indien. University of la Reunion, France.

Lima, E.H.S.M., Felix, M.L., Klefasz, A., Melo, M.T.D., Godfrey, M.H., 2012. From Suriname to Ceará. Green turtle found dead on the coast of Ceará, Brazil. Mar. Turt. Newsl. 135, 18–19.

Lima, E.H.S.M., Lagueux, C.J., Barata, P.C.R., Marcovaldi, M.Â., 2003. Second Record of a Green Turtle (*Chelonia mydas*) Tagged in Brazil and Captured in Nicaragua. Mar. Turt. Newsl. 101, 27.

Lima, E.H.S.M., Melo, M.T.D., Barata, P.C.R., 2010. Incidental capture of sea turtles by the lobster fishery off the Ceará Coast, Brazil. Mar. Turt. Newsl. 128, 16–19.

Lima, E.H.S.M., Melo, M.T.D., Severo, M.M., Barata, P.C.R., 2008. Green Turtle Tag Recovery Further Links Northern Brazil to the Caribbean Region. Mar. Turt. Newsl. 119, 14–15.

Limoges, B., Robillard, M., 1991. Sea turtles in the Bijago archipelago, Guinea-Bissau. Nesting ecology, utilization by man and conservation.

Limpus, C.J., 1971. Sea turtle ocean finding behaviour. Search 2, 385–387.

Limpus, C.J., 1985. A study of the loggerhead turtle, *Caretta caretta*, in eastern Australia. Ph.D. Thesis, Univ. Queensland, Brisbane, Aust. 212 pages 212.

Limpus, C.J., 1993. The green turtle, *Chelonia mydas*, in Queensland: breeding males in the southern Great Barrier Reef. Wildl. Res. 20, 513.

Limpus, C.J., 1995. Global overview of the status of marine turtles: A 1995 viewpoint, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C., pp. 605–609.

Limpus, C.J., 1997. Marine Turtle populations of the Southeast Asia and the western Pacific region: distribution and status., in: Noor, Y.R., Lubis, I.R., Ounsted, R., Troeng, S., Abdullah, A. (Eds.), Proceeding's of the Workshop on Marine Turtle Research and Management in Indonesia. Wetlands International, PHPA, Environment Australia: Bogor., pp. 37–73.

Limpus, C.J., 2002. Western Australia marine turtle review. Western Australian Department of Conservation and Land Management.

Limpus, C.J., 2009. A biological review of Australian marine turtle species. 2. Green turtle, *Chelonia mydas* (Linnaeus). DERM, The State of Queensland. Environmental Protection Agency.

Limpus, C.J., Al-Ghais, S.M., Mortimer, J.A., Pilcher, N.J., 2001. Marine turtles in the Indian Ocean and Southeast Asian region: Breeding, distribution, migration and population trends. Manila, Philippines.

Limpus, C.J., Bell, I., Miller, J.D., 2009. Mixed stocks of green turtles foraging on Clack Reef, Northern Great Barrier Reef identified from long term tagging studies. Mar. Turt. Newsl. 123, 3–5 ST – Mixed stocks of green turtles foraging o.

Limpus, C.J., Chaloupka, M.Y., 1997. Nonparametric regression modelling of green sea turtle growth rates (southern Great Barrier Reef). Mar. Ecol. Prog. Ser. 149, 23–34.

Limpus, C.J., Kamrowski, R.L., 2013. Ocean-finding in marine turtles: the importance of low horizon elevation as an orientation cue. Behaviour 150, 863–893.

Limpus, C.J., Limpus, D.J., Munchow, M., Barnes, P., 2007. Queensland Turtle Conservation Project Raine Island Turtle Study, 2004-2005. Environ. Prot. 2005, 41.

Limpus, C.J., Miller, J.D., Limpus, D.J., Hamann, M., 2002. The Raine Island green turtle rookery : Y2K update, in: Mosier, A., Foley, A., Brost, B. (Eds.), Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-477, pp. 132–134.

Limpus, C.J., Miller, J.D., Parmenter, C.J., Limpus, D.J., 2003. The green turtle, *Chelonia mydas*, population of Raine Island and the Northern Great Barrier Reef: 1843-2001. Mem. Queensl. museum 49, 349–440.

Limpus, C.J., Miller, J.D., Parmenter, C.J., Reimer, D., McLachlan, N., Webb, R., 1992. Migration of green (*Chelonia mydas*) and loggerhead (*Caretta caretta*) turtles to and from eastern Australian rookeries. Wildl. Res. 19, 347–358.

Limpus, C.J., Nicholls, N., 1988. The Southern Oscillation regulates the annual numbers of green turtles (*Chelonia mydas*) breeding around northern Australia. Aust. J. Wildl. Res. 15, 157–161.

Limpus, C.J., Nicholls, N., 2000. ENSO regulation of indo-Pacific green turtle populations, in: Hammer, G.L., Nicholls, N., Mitchell, C. (Eds.), Applications of Seasonal Climate Forecasting in Agricultural and Natural Ecosystems. Kluwer Academic Publications, Dordrecht, The Netherlands, pp. 399–408.

Lindeman, K.C., Snyder, D.B., 1999. Nearshore hardbottom fishes of southeast Florida and effects of habitat burial caused by dredging. Fish. Bull. 97, 508–525.

Lipman, V., Balazs, G.H., 1983. The Lost Hawaiian Island. Honolulu Holiday Annu. 18, 82–87, 150–152.

Lopez, K., 2001. Personal Communication - Yucatan Green Turtles.

López-Barrera, E.A., Longo, G.O., Monteiro-Filho, E.L.A., 2012. Incidental capture of green turtle (*Chelonia mydas*) in gillnets of small-scale fisheries in the Paranaguá Bay, Southern Brazil. Ocean Coast. Manag. 60, 11–18.

López-Mendilaharsu, M., Gardner, S.C., Seminoff, J.A., Riosmena-Rodriguez, R., 2005. Identifying critical foraging habitats of the green turtle (*Chelonia mydas*) along the Pacific coast of the Baja California peninsula, Mexico. Aquat. Conserv. Mar. Freshw. Ecosyst. 15, 259–269.

Lorne, J., Salmon, M., 2007. Effects of exposure to artificial lighting on orientation of hatchling sea turtles on the beach and in the ocean. Endanger. Species Res. 3, 23–30.

Loureiro, N.S., Carvalho, H., Rodrigues, Z., 2011. Praia Grande of Príncipe Island (Gulf of Guinea): an important nesting beach for the green turtle *Chelonia mydas*. Arch. Life Mar. Sci. 28, 89–95.

Louro, C.M.M., Pereira, M.A.M., Costa, A., 2006. The Conservation Status of Marine Turtles in Mozambique. Maputo.

Lucchetti, A., Sala, A., 2009. An overview of loggerhead sea turtle (*Caretta caretta*) bycatch and technical mitigation measures in the Mediterranean Sea. Rev. Fish Biol. Fish. 20, 141–161.

Lutcavage, M.E., Plotkin, P.T., Witherington, B.E., Lutz, P.L.L., 1997. Human impacts on sea turtle survival, in: Lutz, P.L., Musick, J.A. (Eds.), The Biology of Sea Turtles. CRC Press, Boca Raton, Florida, pp. 107–136.

Lwin, M.M., 2009a. Green turtle (*Chelonia mydas*) nesting and conservation activity in Thameehla Island, Myanmar. Indian Ocean Turt. Newsl. 10, 14–18.

Lwin, M.M., 2009b. Tagging Study on Green Turtle (*Chelonia mydas*) at Thameehla Island, Myanmar, in: Proceedings of the 5th International Symposium on SEASTAR2000 and Asian Bio-Logging Science (The 9th SEASTAR2000 Workshop). pp. 15–19.

MacKay, K.T., Petro, G., 2013. Bamboo Bay, Malakula Island, Vanuatau, an important nesting beach for western Pacific green (*Chelonia mydas*) and hawksbill turtles (*Eretmochelys imbricata*), in: Tucker, T., Belskis, L., Panagopoulou, A., Rees, A., Frick, M., Williams, K., LeRoux, R., Stewart, K. (Eds.), Proceedings of the Thirty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NOAA-SEFSC-645, Baltimore, Maryland, p. 182.

Mager, A., 1985. Five-year status reviews of sea turtles listed under the Endangered Species Act of 1973. U.S. Dept. Commerce, NOAA, National Marine Fisheries Service, St. Petersburg, FL. 90 pages.

Maigret, J., 1983. Repartition des tortues de mer sur les cotes ouest Africaines. Bull. Soc. Herp. Fr. 28, 22–34.

Maison, K.A., Kinan-kelly, I., Frutchey, K.P., 2010. Green Turtle Nesting Sites and Sea Turtle Legislation throughout Oceania. U.S. Dep Commerce, NOAA Technical Memorandum NMFS-F/SPO-110, 52 pages.

Makowski, C., Seminoff, J.A., Salmon, M., 2006. Home range and habitat use of juvenile Atlantic green turtles (*Chelonia mydas* L.) on shallow reef habitats in Palm Beach, Florida, USA. Mar. Biol. 148, 1167–1179.

Mancini, A., 2012. Red Sea turtle conservation and research project annual report. HEPCA.

Mancini, A., Koch, V., 2009. Sea turtle consumption and black market trade in Baja California Sur, Mexico. Endanger. Species Res. 7, 1–10.

Mancini, A., Koch, V., Seminoff, J.A., Madon, B., 2012. Small-scale gill-net fisheries cause massive green turtle *Chelonia mydas* mortality in Baja California Sur, Mexico. Oryx 46, 69–77.

Mangel, J.C., Troëng, S., 2001. Report on the 2000 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation and the Ministry of Envirionment, Engery and Telecommunications of Costa rica.

Mann, T.M., 1977. Impact of developed coastline on nesting and hatchling sea turtles in southeastern Florida. Florida Atlantic University, Boca Raton, Florida.

Mansfield, K.L., Seney, E.E., Fagan, M.A., Musick, J.A., Frisch, K.L., Knowles, A.E., 2002. An evaluation of interactions between sea turtles and pound net leaders in the Chesapeake Bay, Virginia. Virginia Institue of Marine Science Final Report submitted to the National Marine Fisheries Service Science Center/Northeast Region. Contract #EA1330-02-SE-0075, Virginia Institute of Marine Science, Gloucester Point, Virginia.

Mansfield, K.L., Wyneken, J., 2013. First satellite tracks of neonate green turtles (*Chelonia mydas*) using a new tag attachment methods, in: Tucker, T., Belskis, L., Panagopoulou, A., Rees, A.F., Frick, M., Williams, K., LeRoux, R., Stewart, K. (Eds.), Proceedings of the Thirty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NOAA NMFS-SEFSC-645, p. 147.

Marcovaldi, M.Â., dei Marcovaldi, G.G., 1999. Marine turtles of Brazil:the history and structure of Projeto TAMAR-IBAMA. Biol. Conserv. 91, 35–41.

Marcovaldi, M.Â., Godfrey, M.H., Mrosovsky, N., 1997. Estimating sex ratios of loggerhead turtles in Brazil from pivotal incubation durations. Can. J. Zool. 75, 755–770.

Marcovaldi, M.Â., Laurent, A., 1996. A six season study of marine turtles nesting at Praia do Forte, Bahia, Brazil, with implications for conservation and management. Chelonian Conserv. Biol. 2, 55–59.

Marcovaldi, M.Â., Patiri, V., Thome, J.C., 2005. Twenty-five years protecting Brazilian sea turtles through a community-based conservation-programme. Mast 3(2) and 4, 39–62.

Marcovaldi, M.A., Sales, G.S., Thomé, J.C.A., Dias Da Silva, A.C.C., Gallo, B.M.G., Lima, E.H.S.M., Lima, E.P., Bellini, C., 2006. Sea turtle and fishery interactions in Brazil: identifying and mitigating potential conflicts. Mar. Turt. Newsl. 112, 4–8.

Marcovaldi, M.Â., Thome, J.C., Sales, G., Coelho, A.C., Gallo, B., Bellini, C., 2002. Brazilian plan for reduction of incidental sea turtle capture in fisheries. Mar. Turt. Newsl. 96, 24–25.

Marcovaldi, N., Gifforni, B.B., Becker, H., Fiedler, F.N., Sales, G., 2009. Sea Turtle Interactions in Coastal Net Fisheries in Brazil, in: Gilman, E. (Ed.), Proceedings of the Technical Workshop on Mitigating Sea Turtle Bycatch in Coastal Net Fisheries. Western Pacific Regional Fishery Management Council, IUCN, Southeast Asian Fisheries Development Center, Indian Ocean – South-East Asian Marine Turtle MoU, U.S. National Marine Fisheries Service, Southeast Fisheries Science Center: Honolulu; Gland, Switze, Honolulu, Hawaii, USA, p. 28.

Marcus, S.J., Maley, C.G., 1987. Comparison of sand temperatures between a shaded and unshaded turtle nesting beach in south Florida, in: Serino, J.L. (Ed.), Presented at Seventh Annual Workshop on Sea Turtle Biology and Conservation, 25-27 February 1987. Wekiwa Springs State Park, p. 16.

Margaritoulis, D., 2007. Status update of marine turtles in the Mediterranean, in: Marine Turtles: Recovery of Extinct Populations. Instituto Canario Ciencias Marinas, pp. 78–94.

Margaritoulis, D., Kousias, N., Nicolopolou, G., Teneketzis, K., 1992. Incidental catch of sea turtles in Greece: the case of Lakonikos Bay, in: Salmon, M., Wyneken, J. (Eds.), Proceedings of the Eleventh Annual Workshop on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-302, Jekyll Island, Georgia, pp. 168–170.

Margaritoulis, D., Panagopoulou, A., 2010. Greece, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Priorities. IUCN, Gland, Switzerland, pp. 85–112.

Margaritoulis, D., Politou, C.-Y., Laurent, L., 2003. Assessing marine turtle bycatch in the trawl fisheries of Greece, in: Margaritoulis, D., Demetropoulos, A. (Eds.), First Mediterranean Conference on Marine Turtles. Barcelona Convention – Bern Convention – Bonn Convention (CMS), Nicosia, Cyprus, pp. 176–180.

Margaritoulis, D., Teneketzis, K., 2003. Identification of a developmental habitat of the green turtle in Lakonikos Bay, Greece, in: Proceedings of the First Mediterranean Conference on Marine Turtles. Barcelona Convention – Bern Convention – Bonn Convention (CMS), pp. 170–175.

Markovina, K., 2008. Ningaloo turtle program annual report 2007-2008. Ningaloo Turtle Program, Exmouth, Western Australia.

Marques, M., da Costa, M.F., de O. Mayorga, M.., Pinheiro, P.R.C., 2004. Water environments: Anthropogenic pressures and ecosystem changes in the Atlantic drainage basins of Brazil. Ambio 33, 68–77.

Márquez-Millán, R., 1990. Sea Turtles of the World. An annotated and illustrated catalog of sea turtle species known to date. FAO Fish. Synopsis No. 125 11, 1–81.

Márquez-Millán, R., 2007. The marine turtles of the eastern coast of Mexico: Abundance, distribution, protection and capture, in: Mast, R.B., Hutchinson, B.J., Hutchinson, A.H. (Eds.), Twenty-Fourth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-567, San Jose, Costa Rica, pp. 138–139.

Márquez-Millán, R., Penaflores S., C., Villanueva O., A., Diaz F., J., 1982. A model for diagnosis of populations of olive ridleys and green turtles of west Pacific tropical coasts, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C., pp. 153–158.

Marsh, H., Kwan, D., 2008. Temporal variability in the life history and reproductive biology of female dugongs in Torres Strait: The likely role of sea grass dieback. Cont. Shelf Res. 28, 2152–2159.

Matsuzawa, Y., 2006. Nesting beach management of eggs and pre-emergent hatchlings of north Pacific loggerhead sea turtles in Japan, in: Kinan, I. (Ed.), Proceedings of the Second Western Pacific Sea Turtle Cooperative Research and Management Workshop. Volume II: North Pacific Loggerhead Sea Turtles. Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, pp. 13–22.

Matsuzawa, Y., Sato, K., Sakamoto, W., Bjorndal, K.A., 2002. Seasonal fluctuatiaons in sand temperature: effects on the incubation period and mortality of loggerhead sea turtle (*Caretta caretta*) pre-emergent hatchlings in Minabe, Japan. Mar. Biol. 140, 639–646.

Maturbongs, J.A., 2000. Marine Turtles Nesting in Sorong, Irian Jaya, Indonesia. Mar. Turt. Newsl. 97, 13.

Mayr, E., 1944. Wallace's Line in the Light of Recent Zoogeographic Studies. Q. Rev. Biol. 19, 1–14.

Mayr, E., 1976. Evolution and the Diversity of Life. Belknap Press of Harvard University Press, Cambridge, Massachusetts.

Mbindo, C., 1996. The status of sea turtle conservation in Madagascar, in: Status of Sea Turtle Conservation in the Western Indian Ocean: Proceedings of the Western Indian Ocean Training Workshop and Strategic Planning Session on Sea Turtles. UNEP Regional Seas Reports and Studies No. 165, IUCN/UNEP, pp. 117–120.

McCauley, S.J., Bjorndal, K.A., 1999. Conservation implications of dietary dilution from debris ingestion: sublethal effects in post-hatchling loggerhead sea turtles. Conserv. Biol. 13, 925–929.

McCoy, M.A., 1997. The traditional and ceremonial use of the green turtle (*Chelonia mydas*) in the Northern Mariana Islands.

McCoy, M.A., 2003. Some aspects of sea turtle interaction with tuna longline fleets in The Federated States of Micronesia.

McCoy, M.A., 2004. Defining parameters for sea turtle research in the Marshall Islands. NOAA Administrative Report AR-PIR-08-04. Silver Spring, Maryland: NOAA National Marine Fisheries Service. Unpublished report.

McCoy, M.A., 2007a. Marshall Islands sea turtle-fisheries interaction outreach and education, phase 2. Final Report to the National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Region, January 2, 2007. 43 pages.

McCoy, M.A., 2007b. Sea turtle bycatch mortality mitigation in the Palau tuna longline fishery. Final Report to the National Oceanic and Atmospheric Administration, National Marine Fisheries Service, Pacific Islands Region, Contract AB133F-05-SE-6409, December 31, 2007. 43 pages.

McCoy, M.A., 2008. Sea turtle bycatch mortality mitigation and outreach: Fiji and Cook Islands longline fisheries. Final report. National Oceanic and Atmospheric Administration Contract Number AB133F07SE3919. Gillbert, Preston and Associates Inc.

McDonald, D., Dutton, P.H., 1990. Fibropapillomas on sea turtles in San Diego Bay, California. Mar. Turt. Newsl. 51, 9–10.

McDonald Dutton, D., Dutton, P.H., 1998. Accelerated growth in San Diego Bay green turtles?, in: Epperly, S.P., Braun, J. (Eds.), Proceedings of the Seventeenth Annual Sea Turtle Symposium. NOAA Technical Memorandum NMFS-SEFSC-415, pp. 175–176.

McElhany, P., Ruckelshaus, M.H., Ford, M.J., Wainwright, T.C., Bjorkstedt, E.P., 2000. Viable Salmonid Populations and the Recovery of Evolutionarily Significant Units. U.S. Dept. Commerce, NOAA Tech. Memo. NMFS-NWFSC-42.

McGowan, A., Broderick, A.C., Deeming, J., Godley, B.J., Hancock, E.G., 2001. Dipteran infestation of loggerhead (*Caretta caretta*) and green (*Chelonia mydas*) sea turtle nests in northern Cyprus. J. Nat. Hist. 35, 573–581.

Mckenzie, C., Godley, B.J., Furness, R.W., Wells, D.E., 1999. Concentrations and patterns of organochlorine contaminants in marine turtles from Mediterranean and Atlantic waters. Mar. Environ. Res. 47, 117–135.

McMichael, E., Norem, A., Carthy, R.R., Summers, T., 2006. Summary of 2003 cold stun turtles in St Joseph Bay, Florida, in: Pilcher, N.J. (Ed.), Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-536, Kuala Lumpur, Malaysia, pp. 184–186.

McTaggart-Cowan, R., Bosart, L.F., Davis, C.A., Atallah, E.H., Gyakum, J.R., Emanuel, K.A., 2006. Analysis of Hurricane Catarina (2004). Mon. Wea. Rev. 134, 3029–3053.

Mediterranean Association to Save the Sea Turtles, 2013. Kazanli, Turkey [WWW Document]. URL http://www.medasset.org/cms/index.php?option=com_content&view=article&id=46&Itemid=39&lang=en (accessed 2.21.13).

Meffe, G.K., Carroll, C.R., Pimm, S.L., 1994. Community level conservation: species interactions, disturbance regimes, and invading species, in: Meffe, G.K., Carroll, C.R. (Eds.), Principles of Conservation Biology. Sunderland: Sinaeur Associates, Inc, p. 600.

Mendonça, M.T., 1981. Comparative growth rates of wild immature *Chelonia mydas* and *Caretta caretta* in Florida. J. Herpetol. 15, 447–451.

Mendonça, M.T., Ehrhart, L.M., 1982. Activity, population size and structure of immature *Chelonia mydas* and *Caretta caretta* in Mosquito Lagoon, Florida. Copeia 1, 161–167.

Mendonça, V.M., Saady, S. Al, Al-Kiyumi, A.A., Erzini, K., 2010. Interactions between Green Turtles (*Chelonia mydas*) and Foxes (Vulpes vulpes arabica, V. rueppellii sabaea, and V. cana) on Turtle Nesting Grounds in the Northwestern Indian Ocean: Impacts of the Fox Community on the Behavior of Nesting Sea Turtles at the. Zool. Stud. 49, 437–452.

Mendoza-Salgado, R.A., Amador Silva, E.S., Lechuga-Devéze, C.H., Martínez, J.A., 2003. Muerte masiva de fauna marina en Bahía Magadalena (Massive Death of Marine Fauna in Magdalena Bay). Rev. Biol. 17, 64–68.

Meschini, P., 1997. Prima segnalazione di una cattura accidentale di *Chelonia mydas* (Linneo, 1758) lungo il litorale livornese (secche della Meloria), in: Fauna Del Mediterraneo. Immagini E Note Di Ecologia Marina. Quaderni Dell'acquario 3. Acquario di Livorno, Livorno, pp. 5–11.

Meylan, A.B., Bowen, B.W., Avise, J.C., 1990. A genetic test of the natal homing versus social facilitation models for green turtle migration. Science (80-.). 248, 724–727.

Meylan, A.B., Meylan, P.A., Ordoñez Espinosa, C., 2013. Sea turtles of Bocas del Toro Province and the Comarca Ngobe-Bugle, Republic of Panama. Chelonia Conserv. Biol. 12, 17–33.

Meylan, A.B., Sadove, S., 1986. Cold-stunning in Long Island Sound, New York. Mar. Turt. Newsl. 37, 7–8.

Meylan, A.B., Witherington, B.E., Brost, B., Rivero, R., Kubilis, P.S., 2006. Sea turtle nesting in Florida, USA: Assessments of abundance and trends for regionally significant populations of Caretta, Chelonia, and Dermochelys, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Book of Abstracts' Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservatio. ISTS, Athens, Greece, pp. 306–307.

Meylan, P.A., Meylan, A.B., 2011. The Ecology and Migrations of Sea Turtles 8. Tests of the Developmental Habitat Hypothesis. Bull. Am. Museum.

Miller, J.D., 1989. Marine turtles: Vol. 1: An assessment of the conservation status of marine turtles in the Kingdom of Saudi Arabia. MEPA Coast. Mar. Manag. Ser. pp.1-209. Minist. Def. Aviat. Kingdom Saudi Arab. Rep. No. 9.

Miller, J.D., Limpus, C.J., Godfrey, M.H., 2003. Nest site selection, oviposition, eggs, development, hatchling, and emergence of loggerhead turtles, in: Bolten, A.B., Witherington, B.E. (Eds.), Loggerhead Sea Turtles. Smithsonian Books, Washington D.C., pp. 125–143.

Miller, J.D., Lutz, P.L., Musick, J.A., 1997. Reproduction in sea turtles, in: Lutz, P.L., Musick, J.A. (Eds.), Biology of Sea Turtles, Volume I. CRC Press. Inc, Boca Raton, Florida, pp. 51–81.

Miller, M.H., Carlson, J., Cooper, P., Kobayashi, D., Nammack, M., Wilson, J., 2013. Status review report: scalloped hammerhead shark (Sphyrna lewini). Report to National Marine Fisheries Service, Office of Protected Resources.

Milliken, T., Tokunaga, H., 1987. The Japanese Sea Turtle Trade, 1970-1986, Education. Center for Environmental Education.

Milton, S.L., Lutz, P.L., 2003. Physiological and genetic responses to environmental stress, in: Lutz, P.L., Musick, J.A., Wyneken, J. (Eds.), The Biology of Sea Turtles, Volume II. CRC Press, Boca Raton, Florida, pp. 163–197.

Ministry of Fisheries, 2003. Conservation Satus Report Marine Turtles and Their Habitats in Viet Nam. IUCN.

Ministry of Fisheries, 2004. Marine turtle conservation action plan until 2010 in Viet Nam. Viet Nam Ministry of Fisheries. Decision 175/QD-BTS. 8 March 2004. Ha Noi.

Minton, S.A., 1966. A contribution to the herpetology of West Pakistan. Bull. Am. Museum Nat. Hist. 134.

Miyawaki, I., Kamezaki, N.I., Kolinski, S.P., 2000. A migration of a green turtle from Yap Islands to Muroto, Kochi Prefecture, Japan. Umigame Newsl. Japan 45, 13.

Mobaraki, A., 2004. Green turtle nesting on the Gulf of Oman coastline of the Islamic Republic of Iran. Mar. Turt. Newsl. 104, 11.

Mobaraki, A., 2007. Sea turtle study in Iranian coasts of Oman Sea, in: Mast, R.B., Hutchinson, B.J., Hutchinson, A.H. (Eds.), Proceedings of the Twenty-Fourth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-567, p. 101.

Mobaraki, A., 2011. Sea turtle situation, studies and conservation in the Islamic Republic of Iran [WWW Document]. Indian Ocean - South-East Asian Mar. Turt. Memo. Underst. Profile Mon. URL http://www.ioseaturtles.org/pom_detail.php?id=109

Moncada, F.G., Abreu-Grobois, F.A., Muhlia-Melo, A., Bell, C.D., Troëng, S., Bjorndal, K.A., Bolten, A.B., Meylan, A.B., Zurita, J.C., Espinosa, G., others, 2006. Movement patterns of green turtles (*Chelonia mydas*) in Cuba and adjacent Caribbean waters inferred from flipper tag recaptures. J. Herpetol. 40, 22–34.

Moncada, F.G., Azanza, J., Nodarse, G., Medina, Y., Forneiro, Y., Gerhartz, J.L., 2011. Informe anual monitoreo tortugas marinas, Cuba. Grupo Nacional para la conservacion de las tortugas marinas en Cuba.

Moncada, F.G., Forneiro, Y., Azanza, J., Nodarse, G., Medina, Y., Blanco, R., Valido, L., Gerhartz, J., 2010. Informe anual de investigacion y conservacion de las tortugas marinas. Grupo Nacional para la conservacion de las tortugas marinas en Cuba.

Moncada, F.G., Prieto, A., 2000. Incidence of Fibropapillomas in the green turtle (*Chelonia mydas*) in Cuban waters, in: Kalb, H.J., Wibbels, T. (Eds.), Proceedings of the Nineteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-443. 294 pages, pp. 40–41.

Montes-Osorio, N.L., Jolon-Morales, M.R., Ruiz-Fumagalli, R., Sanchez-Castaneda, R., 2007. Relative abundance of nesting marine turtles on the coasts of Guatemala from July to December 2002, in: Mast, R.B., Hutchinson, B.J., Hutchinson, A.H. (Eds.), Proceedings of the Twenty-Fourth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-567. 205 pages, p. 101.

Monzón-Argüello, C., Lopez-Jurado, L.F., Rico, C., Marco, A., Lopez, P., Hays, G.C., Lee, P.L.M., 2010. Evidence from genetic and Lagrangian drifter data for transatlantic transport of small juvenile green turtles. J. Biogeogr. 37, 1752–1766.

Moreira, L.M.P., Baptistotte, C., Scalfone, J., Thomé, J.C.A., Almeida, A.P., 1995. Occurrence of *Chelonia mydas* on the Island of Trindade, Brazil. Mar. Turt. Newsl. 70, 2.

Moreira, L.M.P., Bjorndal, K.A., 2006. Estimates of green turtle (*Chelonia mydas*) nests on Trindade Island, Brazil, South Atlantic, in: Pilcher, N.J. (Ed.), Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-536, Kuala Lumpur, Malaysia, p. 174.

Moritz, C., 1994. Defining "Evolutionarily Significant Units" for conservation. Trends Ecol. Evol. 9, 373–375.

Moritz, C., Broderick, D., Dethmers, K.E.M., FitzSimmons, N.N., Limpus, C.J., 2002. Population genetics of Southeast Asian and Western Pacific green turtles, *Chelonia mydas*. Final Report to UNEP/CMS.

Morreale, S.J., Meylan, A.B., Sadove, S.S., Standora, E.A., 1992. Annual occurrence and winter mortality of marine turtles in New York waters. J. Herpetol. 26, 301–308.

Mortimer, J.A., 1981. The Feeding Ecology of the West Caribbean Green Turtle (*Chelonia mydas*) in Nicaragua. Biotropica 13, 49–58.

Mortimer, J.A., 1984. Marine turtles in the Republic of the Seychelles: status and management, report on project 1809 (1981-1984). Gland, Switzerland.

Mortimer, J.A., 1985. Recovery of green turtles on Aldabra. Oryx 19, 146–150.

Mortimer, J.A., 1991a. Recommendations for the Management of Turtle Islands Park, Sabah. WWF Malaysia.

Mortimer, J.A., 1991b. Marine turtle populations of Pulau Redang: their status and recommendations for their management.

Mortimer, J.A., 1992. Marine turtle conservation in Malaysia. Malayan Nat. J. 45, 353–361.

Mortimer, J.A., 2012. Seasonality of Green Turtle (*Chelonia mydas*) Reproduction at Aldabra Atoll, Seychelles (1980–2011) in the Regional Context of the Western Indian Ocean. Chelonian Conserv. Biol. 11, 170–181.

Mortimer, J.A., Carr, A.F., 1987. Reproduction and Migrations of the Ascension Island Green Turtle (*Chelonia mydas*). Copeia 103–113.

Mortimer, J.A., Collie, J., 1998. Status and conservation of sea turtles in the Republic of Seychelles, in: Proceedings of the 17th Annual Sea Turtle Symposium. NOAA Tech. Memor. NMFS-SEFSC-415, pp. 70–72.

Mortimer, J.A., Collie, J., Mbindo, C., 1996. The Status of sea turtle conservation in the Republic of Seychelles, in: Humphrey, S.L., Salm, R.V. (Eds.), Status of Sea Turtle Conservation in the Western Indian Ocean: Proceedings of the Western Indian Ocean Training Workshop and Strategic Planning Session on Sea Turtles. Regional Seas Reports and Studies . No. 165. UNEP, Sodwana Bay, South Africa, pp. 103–115.

Mortimer, J.A., Day, M., 1999. Sea turtle populations and habitats in the Chagos Archipelago. Ecol. Chagos Arch. Linn. Soc. Occas. Publ. 2, 159–176.

Mortimer, J.A., Von Brandis, R.G., Liljevik, A., Chapman, R., Collie, J., 2011. Fall and rise of nesting green turtles (*Chelonia mydas*) at Aldabra Atoll, Seychelles: Positive response to four decades of protection (1968–2008). Chelonian Conserv. Biol. 10, 165–176.

Mosier, A., 1998. The impact of coastal armoring structures on sea turtle nesting behavior at three beaches on the East Coast of Florida. University of South Florida, Tampa, Florida.

Mosier, A.E., Witherington, B.E., 2002. Documented effects of coastal armoring structures on sea turtle nesting behavior, in: Mosier, A., Foley, A., Brost, B. (Eds.), Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFC-477, pp. 304–306.

Mrosovsky, N., 1968. Nocturnal emergence of hatchling sea turtles: control by thermal inhibition of activity. Nature 220, 1338–1339.

Mrosovsky, N., 1980. Thermal Biology of Sea Turtles. Am. Zool. 20, 531–547.

Mrosovsky, N., Dutton, P.H., Whitmore, C.P., 1984. Sex ratios of two species of sea turtle nesting in Suriname. Can. J. Zool. 62, 2227–2239.

Mrosovsky, N., Yntema, C.L., 1980. Temperature dependence of sexual differentiation in sea turtles: implications for conservation practices. Biol. Conserv. 18, 271–280.

Mugetti, A., Calcagno, A., Brieva, C., Giangiobbe, M., Pagani, A., Gonzalez, S., 2004. Aquatic habitat modifications in La Plata River basin, patagonia and associated marine areas. Ambio 33, 78–87.

Muhara, R., Herlina, D., 2012. Green turtle conservation in Pangumbahan: lesson learned from Sukabumi, in: East Asian Seas Congress. Changwong City, Republic of Korea, p. 83.

Muir, C.E., 2005. The status of marine turtles in the United Republic of Tanzania, East Africa. Sea Sense Report (Tanzania Turtle and Dugong Conservation Programme).

Murdoch, T.J.T., A.F., G., Outerbridge, M., Ward, J., Manuel, S., Gray, J., Nash, A., Coates, K.A., Pitt, J., Fourqurean, J.W., Barnes, P.A., Vierros, M., Holzer, K., Smith, S.R., 2007. Large-scale decline in offshore seagrass meadows in Bermuda. Mar. Ecol. Prog. Ser. 339.

Murman, M.I., 2011. El impacto de los desechos antrópicos en individuos juveniles de *Chelonia mydas* (tortuga verde), Cerro Verde, Uruguay. Universidad CAECE.

Musick, J.A., Limpus, C.J., 1997. Habitat utilization and migration in juvenile sea turtles, in: Lutz, P.L., Musick, J.A. (Eds.), The Biology of Sea Turtles. CRC Press, pp. 137–163.

Myers, N., Mittermeier, R.A., Mittermeier, C.G., da Fonseca, G.A.B., Kent, J., 2000. Biodiversity hotspots for conservation priorities. Nature 403, 853–858.

Nada, M., Casale, P., 2008. Marine turtles in the Mediterranean Egypt: threats and conservation priorities. Rome, Italy.

Nada, M., Casale, P., 2010. Egypt, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Priorities. IUCN, Gland, Switzerland, pp. 65–76.

Nada, M., Casale, P., 2011. Sea turtle bycatch and consumption in Egypt threatens Mediterranean turtle populations. Oryx 45, 143–149.

Nada, M.A., 2001. Observations on the trade in sea turtles at the fish market of Alexandria, Egypt. Zool. Middle East 24, 109–118.

Nalo-Ochona, C.M., 2000. Histopathology and histochemistry of fibropapilloma on the carapce of green turtles (*Chelonia mydas*) in the Baguan Island Marine Turtle Sanctuary. Master of Science Thesis, University of the Philippines Los Banos.

Namboothri, N., Swaminathan, A., Shanker, K., 2012. A compilation of data from Satish Bhaskar's sea turtle surveys of the Andaman and Nicobar Islands. Indian Ocean Turt. Newsl. 16, 4–13.

Naro-Maciel, E., Becker, J.H., Lima, E.H.S.M., Marcovaldi, M.Â., DeSalle, R., 2007. Testing Dispersal Hypotheses in Foraging Green Sea Turtles (*Chelonia mydas*) of Brazil. J Hered 98, 29–39.

Naro-Maciel, E., Bondioli, a. C. V., Martin, M., de Padua Almeida, A., Baptistotte, C., Bellini, C., Marcovaldi, M.Â., Santos, a. J.B., Amato, G., 2012. The Interplay of Homing and Dispersal in Green Turtles: A Focus on the Southwestern Atlantic. J. Hered. 103, 792–805.

Naro-Maciel, E., Le, M., FitzSimmons, N.N., Amato, G., 2008. Evolutionary relationships of marine turtles: A molecular phylogeny based on nuclear and mitochondrial genes. Mol. Phylogenet. Evol. 49, 659–662.

Naro-Maciel, E., Mrosovsky, N., Marcovaldi, M.Â., 1999. Thermal profiles of sea turtle hatcheries and nesting areas at Praia do Forte, Brazil. Chelonian Conserv. Biol. 3, 407–413.

Naser, H.A., 2012. Evaluation of the environmental impact assessment system in Bahrain. J. Environ. Prot. (Irvine,. Calif). 3, 233–239.

Nasome, E. in consultation with Tokaduadua, E. and Batibasaga, A., 2010. Review of Marine Turtles Legislation in Fiji. Secretariat of the Pacfic Regional Environment Programme (SPREP). Apia, Samoa.

National Fisheries Authority Papua New Guinea, 2007. A review of fisheries and marine resources in Morobe Province, Papua New Guinea. National Fisheries Authority and Coastal Fisheries Management and Development Project.

National Research Council, 1990a. Managing coastal erosion. National Academy Press; Washington D.C.

National Research Council, 1990b. Decline of the sea turtles: causes and prevention. National Academy Press; Washington D.C.

National Research Council, 1990c. Sea turtle mortality associated with human activities, in: Decline of the Sea Turtles: Causes and Prevention. National Academy ofPres, pp. 74–117.

National Research Council, 1996. An assessment of techniques for removing offshore structures. National Academy Press; Washington D.C.

National turtle management and conservation in Brunei Darussalam [WWW Document], 2010. URL http://www.docstoc.com/docs/44766009/National-Turtle-Management-and-Conservation-in-Brunei-Darussalam

Naughton, J.J., 1991. Sea Turtle Survey at Oroluk Atoll and Minto Reef, Federated States of Micronesia. Mar. Turt. Newsl. 55, 9–12.

Nelson, D.A., 1988. Life history and environmental requirements of loggerhead turtles. U.S. Fish Wildl. Serv. Biol. Rep. 88(23). Tech. Rep. EL-86-2 (Rev.). U.S. Army Corps Eng.

Nelson, D.A., Mauck, K., Fletemeyer, J.R., 1987. Physical effects of beach nourishment on sea turtle nesting, Delray Beach, Florida.

Nelson, D.A., Shafer, D.J., 1996. Effectiveness of a sea turtle-deflecting hopper dredge draghead in Port Canaveral Entrance Channel, Florida. U. S. Army Corps of Engineers, Dredging Operations Technical Support Program, Miscellaneous Paper D-96-3.

Nelson Sella, K., Salmon, M., Witherington, B.E., 2006. Filtered streetlights attract hatcling marine turtles. Chelonia Conserv. Biol. 5, 255–261.

Nicholls, R.J., Cazenave, A., 2010. Sea-level rise and its impact on coastal zones. Science (80-.). 328, 1517–1520.

Nichols, W.J., 2003. Biology and conservation of sea turtles in Baja California, Mexico. Ph.D. Thesis. School of Renewable and Natural Resources, The University of Arizona, Tucson, Arizona.

Nichols, W.J., Aridjis, H., Hernandez, A., Machovina, B., Villavicencios, J., 2002. Black market sea turtle trade in the Californias. WiLDCOAST.

Niethammer, K.R., Balazs, G.H., Hatfield, J.S., Nakai, G.L., Megyesi, J.L., 1997. Reproductive biology of the green turtle (*Chelonia mydas*) at Tern Island, French Frigate Shoals, Hawaii. Pacific Sci. 51, 36–47.

Niethammer, K.R., Balazs, G.H., Nakai, G.L., McDermond, D.K., 1992. Great frigatebird (Fregata minor) predation: not a factor in hatchling green turtle (*Chelonia mydas*) survival at French Frigate Shoals, Hawaii. Colon. Waterbirds 15, 128–131.

Nishizawa, H., Abe, O., Okuyama, J., Kobayashi, M., Arai, N., 2011. Population genetic structure and implications for natal philopatry of nesting green turtles *Chelonia mydas* in the Yaeyama Islands, Japan. Endanger. Species Res. 14, 141–148.

NMFS, 2002. Biological opinion on shrimp trawling in the Southeastern United States under the sea turtle conservation regulations and as managed by the fishery management plans for shrimp in the South Atlantic and Gulf of Mexico.

NMFS, 2005a. Recovery Plan for the North Atlantic Right Whale (Eubalaena glacialis). National Marine Fisheries Service, Silver Spring, MD.

NMFS, 2005b. Biological opinion on the Hawaii-based pelagic, deep-sea longline fishery, October 4, 2005. National Marine Fisheries Service, Silver Spring, MD.

NMFS, 2010. Recovery Plan for the fin whale (*Balaenoptera physalus*). National Marine Fisheries Service, Silver Spring, MD.

NMFS, 2011. Final Recovery Plan for the Sei Whale (*Balaenoptera borealis*). National Marine Fisheries Service, Office of Protected Resources, Silver Spring, MD.

NMFS, 2012. Biological opinion on the continued operation of the Hawaii-based shallow-set longline swordfish fishery - under Amendement 18 to the Fishery Management Plan for Pleagic Fisheries of the Western Pacific region. Pacific Islands Regional Office. January 30.

NMFS and USFWS, 1998. Recovery Plan for U.S. Pacific Populations of the Green Turtle (*Chelonia mydas*). National Marine Fisheries Service, Silver Spring, MD.

NMFS and USFWS, 2007. Green sea turtle (*Chelonia mydas*) 5-year review: summary and evaluation. NMFS Office of Protected Resources, Silver Spring and USFWS Southeast Region, Jacksonville.

NMFS and USFWS, 2008. Recovery Plan for the Northwest Atlantic Population of the Loggerhead Sea Turtle (*Caretta caretta*), Second Revision. National Marine Fisheries Service, Silver Spring, MD.

Nodarse, G., Moncada, F.G., Medina, Y., Rodríguez, C., Hernández, F., Blanco, R., Escobar, E., 2010. Comportamiento de la anidación de tortugas marinas en los Cayos San Felipe y Archipiélago de los Canarreos, Cuba (2001-2006). Rev. Cuba. Investig. Pesq. 27.

Nodarse, G., Moncada, F.G., Menesis, A., Rodriguez, C., 2000. Long-term monitoring of nesting of the green sea turtle (*Chelonia mydas*) in the southwest platform of Cuba, in: Abreu-Grobois, A., Briseño Dueñas, R., Marquez-Millan, R., Sarti-Martinez, L. (Eds.), Proceedings of the Eighteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Tech Memo NMFS SEFSC-436, pp. 68–69.

Nolasco del Aguila, D., Debade, X., Harrison, E., 2009. Report on the 2008 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation and the Ministry of Envirionment, Engery and Telecommunications of Costa rica.

Obermeier, A., 2002. Conservation programme on Sangalaki Island, East Kalimantan, Indonesia. Mar. Turt. Newsl. 96, 27–28.

Obura, D., Stone, G., 2002. Phoenix Islands Summary of Marine and Terrestrial Assessments Conducted in the Republic of Kiribati, June 5-July 10, 2002. Primal Ocean Project Technical Report, Boston, MA.

Oceanic Fisheries Programme, 2001. A review of turtle by-catch in the western and central Pacific Ocean tuna fisheries.

Ogren, L.H., 1989. Green turtle (*Chelonia mydas*): status report on the green turtle., in: Ogren, L., Berry, F., Bjorndal, K., Kumpf, H., Mast, R., Medina, G., Reichart, H., Witham, R. (Eds.), Proceedings of the Second Western Atlantic Turtle Symposium. NOAA Technical Memorandum NMFS-SEFC-226, pp. 89–94.

Okemwa, G.M., 2003. Nesting and mortality patterns of sea turtles along the Kenyan coast (1997-2000), in: Mombasa, Kenya, Kenya Sea Turtle Conservation Committee (KESCOM).

Okemwa, G.M., Nzuki, S., Mueni, E.M., 2004. The status and conservation of sea turtles in Kenya. Mar. Turt. Newsl. 105, 1–6.

Okemwa, G.M., Wamukota, A., 2006. An overview of the status of green turtles (*Chelonia mydas*) in Kenya, in: Frick, M., Panagopoulou, A., Rees, A.. F., Williams, K. (Eds.), Book of Abstracts. Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece, p. 311.

Oleson, E.M., Boggs, C.H., Forney, K.A., Hanson, M.B., Kobayashi, D.R., Taylor, B.L., Wade, P.R., Ylitalo, G.M., 2010. Status review of Hawaiian insular false killer whales (Pseudorca crassidens) under the Endangered Species Act. U.S. Department of Commerce, NOAA Technical Memorandum NMFS-PIFSC-22.

Ono, R., Addison, D.J., 2009. Ethnoecology and Tokelauan fishing lore from Atafu Atoll, Tokelau. SPC Traditional Marine Resource Management and Knowledge Information Bulletin #26.

Orós, J., Torrent, a, Calabuig, P., Déniz, S., 2005. Diseases and causes of mortality among sea turtles stranded in the Canary Islands, Spain (1998-2001). Dis. Aquat. Organ. 63, 13–24.

Orrego, C.M., Arauz, R., 2004. Mortality of sea turtles along the Pacific Coast of Costa Rica, in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Tech Memo NMFS-SEFSC-528, pp. 265–266.

Oruç, A., 2001. Trawl fisheries in the eastern Mediterranean and their impact on marine turtles. Zool. Middle East 119–125.

Ottenwalder, J.A., 1981. Estudio preliminar sobre el status, distribucion, y biologia reproductiva de las tortugas marinas en la Republica Dominicana. Report prepared for the Western Atlantic Sea Turtle Symposium and presented to the National Marine Fisheries Service and Caribbean Conservation Corporation.

Ottenwalder, J.A., 1987. Ad Hoc National Report to WATS II for Haiti, in: Prepared for the Second Western Atlantic Turtle Symposium (WATS II), 12-16 October 1987, Mayaguez, Puerto Rico. Doc. 044, p. 29.

Oz, M., Erdogan, A., Kaska, Y., Duşen, S., Aslan, A., Sert, H., Yavuz, M., Tunç, M., 2004. Nest temperatures and sex-ratio estimates of loggerhead turtles at Patara beach on the southwestern coast of Turkey. Can. J. Zool. 82, 94–101.

Pacific News Center, 2012. SPC: Kiribati to end beach mining for aggregates. News Release, Wednesday, 19 September 2012 [WWW Document]. URL http://www.pacificnewscenter.com/index.php?option=com_content&view=article&id=27367:spc-kiribati-to-end-beach-mining-for-aggregates&catid=45:guam-news&Itemid=156 (accessed 4.18.13).

Palacios, A.I., 2012a. Sea turtle stock and nesting assessment in the CNMI. Commonwealth of the Northern Mariana Islands, Department of Lands and Natural Resources, Saipan, CNMI.

Palacios, A.I., 2012b. CNMI sea turtle program annual report, January 1, 2011 - December 31, 2011. Commonwealth of the Northern Mariana Islands, Department of Lands and Natural Resources, Saipan, CNMI, Saipan, CNMI.

Palaniappan, P.M., Mahedi, P.A., Lee, C.N.Y., Day, K.V., Day, P.S., 2004. Turtles and people: A case study of Sipadan (Sabah, Malaysia), in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-528, p. 368.

Palau Bureau of Marine Resources, 2008. Marine turtle conservation and monitoring program, final report. Bureau of Marine Resources, Republic of Palau. Unpublished grant report. 31 pp.

Palomares, M.L., Pauly, D., 2004. West African Marine Ecosystems: Models and Fisheries Impacts. Fish. Cent. Res. Rep. 12, 221 p.

Palomares, M.L.D., Heymans, J.J., Pauly, D., 2007. Historical ecology of the Raja Ampat Archipelago, Papua Province, Indonesia. Hist. Philos. Life Sci. 29, 33–56.

Panagopoulos, D., Sofouli, E., Teneketzis, K., Margaritoulis, D., 2003. Stranding data as an indicator of fisheries induced mortality of sea turtles in Greece, in: First Mediterranean Conference on Marine Turtles. Barcelona Convention – Bern Convention – Bonn Convention (CMS), Nicosia, Cyprus, pp. 202–206.

Panagopoulou, A., Koutsodendris, A., Margaritoulis, D., 2008. Interactions with small scale fisheries in Greece: an important factor for the reduction of sea turtle mortality at sea, in: Proceedings of the Twenty-Seventh Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-569, p. 186.

Papathanasopoulou, N., 2010. Kuwait turtle nesting season 2009 [WWW Document]. Bazaar Mag. - Kuwait. URL http://www.bazaar-magazine.com/baz/bazaar/index.php?show=eIndex&show_filter=view&action=article&art_id=ART00000000644 (accessed 4.2.12).

Paris, B., Agardy, T., 1993. La tortue verte et la tortue olive de ridley de l'Archipel des Bijagos: Identification de leur importance dans le contexte mondial et contribution a la proposition de zonage d'une reserve de la biosphere.

Parker, D.M., Dutton, P.H., Balazs, G.H., 2011. Oceanic Diet and Distribution of Haplotypes for the Green Turtle, *Chelonia mydas* , in the Central North Pacific. Pacific Sci. 65, 419–431.

Parsons, J.J., 1962. The green turtle and man. University of Florida Press, Gainesville, Florida.

Passfield, K., 1998. A report of a survey of the marine resources of Fakaofo Atoll, Tokelau. Fisheries and Environmental Resource Consultants Ltd.

Patino-Martinez, J., Marco, A., Quiñones, L., Hawkes, L.A., 2012. A potential tool to mitigate the impacts of climate change to the Caribbean leatherback sea turtle. Glob. Chang. Biol. 18, 401–411.

Patrício, A.R., Velez-Zuazo, X., Diez, C., Dam, R., Sabat, A., 2011. Survival probability of immature green turtles in two foraging grounds at Culebra, Puerto Rico. Mar. Ecol. Prog. Ser. 440, 217–227.

Patrício, R., Diez, C.E., van Dam, R.P., 2014. Spatial and temporal variability of immature green turtle abundance and somatic growth in Puerto Rico. Endanger. Species Res. 23, 51–62.

Pavlakis, P., Sieber, A., Alexandry, S., 1996. Monitoring oil-spill pollution in the Mediterranean with ERS SAR. Eur. Sp. Agency EOQ 52.

Pawikan Conservation Project, 2013. Annual Accomplishment Reports, 1984-2013. Pawikan Conservation Project, Wildlife Resources Division, Biodiversity Management Bureau, Department of Environment and Natural Resources (DENR-BMB-WRD-PCP).

Pearse, D.E., Eckerman, C.M., Janzen, F.J., Avise, J.C., 2001. A genetic analogue of "mark-recapture" methods for estimating population size: an approach based on molecular parentage assessments. Mol. Ecol. 10, 2711–2718.

Peckham, S.H., Maldonado Diaz, D., Walli, A., Ruiz, G., Crowder, L.B., Nichols, W.J., 2007. Small-scale fisheries bycatch jeopardizes endangered Pacific loggerhead turtles. PLoS One e1041.

Pena Mosquera, M., Baquero Gallegos, A., Munoz Perez, J., Puebla Jimenez, F., Alvarez, V., Chalen Norona, X., 2009. El Parque National Machalilla: zona critica de anidacion para la tortuga carey (*Eretmochelys imbricata*) y verde (*Chelonia mydas*) en el Ecuador y el Pacific. Temporadas 2007-2009 Oriental.

Peñate, J.G., Karamoko, M., Bamba, S., Djadji, G., 2007. An Update on Marine Turtles in Côte d ' Ivoire , West Africa. Mar. Turt. Newsl. 2001–2002.

Pendoley, K., 2000. The influence of gas flares on the orientation of green turtle hatchlings at Thevenard, Western Australia, in: Pilcher, N.J., Ismail, G. (Eds.), Sea Turtles of the Indo-Pacific: Research, Management and Conservation. ASEAN ACADEMIC Press, Ltd., London, pp. 130–142.

Pendoley, K.L., 2005. Sea turtles and the environmental management of industrial activities in north west Western Australia. Murdoch University.

PERSGA/GEF, 2001. Country Reports. Strategic Action Programme for the Red Sea and Gulf of Aden. PERSGA, Jeddah and World Bank, Washington, D.C. 205 pp.

PERSGA/GEF, 2004. Regional action plan for the conservation of marine turtles and their habitats in the Red Sea and Gulf of Aden, Development. PERSGA, Jeddah.

Pesca, S. de, 1990. Fishery closure for green turtle protection. D. Of. la Fed. 22, 21–24.

Peterson, S., 2005. Initial bycatch assessment: South Africa's domestic longline fisher, 2000-2003. BirdLife Africa 45.

Peterson, S., Nel, D., Omardien, A., 2007. Towards and ecosystem approach to longline fisheries in the Benguela: an assessment of impacts on seabirds, sea turtles and sharks.

Peterson, S.L., Honig, M.B., Ryan, P.G., Nel, R., Underhill, L.G., 2009. Turtle bycatch in the pelagic longline fishery off southern Africa. African J. Mar. Sci. 31, 87–96.

Petit, M., 2009. Rapport final relatif au suivi de sites de ponte de tortues sur L'Atoll de Tetiaroa. Moorea, French Polynesia.

Petit, M., 2011. Tetiaroa's nesting survey campaign: 4 years of monitoring.

Philip, M., 2002. Marine turtle conservation in Papua New Guinea, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 143–146.

Pianka, E.R., 1974. Evolutionary Ecology. Harper and Row, New York.

Pilcher, N.J., 1999. Cement dust pollution as a cause of sea turtle hatchling mortality at Ras Baridi, Saudi Arabia. Mar. Pollut. Bull. 38, 966–969.

Pilcher, N.J., 2000. The green turtle, *Chelonia mydas*, in the Saudi Arabian Gulf. Chelonian Conserv. Biol. 3, 730–734.

Pilcher, N.J., 2006. Status of sea turtles in Qatar. IEM Tech Report.

Pilcher, N.J., 2010. Population structure and growth of immature green turtles at Mantanani, Sabah, Malaysia. J. Herpetol. 44, 168–171.

Pilcher, N.J., Al-Merghani, M., 2000. Reproductive biology of green turtles at Ras Baridi, Saudi Arabia. Herpetol. Rev. 31, 142–147.

Pilcher, N.J., Basintal, P., 2000. Reproductive biology of the green turtle *Chelonia mydas* in Sabah, Malaysia. Asian J. Trop. Biol. 4, 59–66.

Pintus, K.J., Godley, B.J., McGowan, A., Broderick, A.C., 2009. Impact of Clutch Relocation on Green Turtle Offspring. J. Wildl. Manage. 73, 1151–1157.

Pita, J., Broderick, D., 2005. Hawksbill turtles in the Solomon Islands, in: Kinan, I. (Ed.), Proceedings of the Second Western Pacific Sea Turtle Cooperative Research and Management Workshop. Volume I: West Pacific Leatherback and Southwest Pacific Hawksbill Sea Turtles. Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 101–102.

Plotkin, P., 2003. Adult migrations and habitat use, in: Lutz, P., Musick, J., Wyneken, J. (Eds.), The Biology of Sea Turtles, Volume II. CRC Press, Boca Raton, Florida, pp. 225–242.

Plotkin, P., Amos, A.F., 1990. Effects of anthropogenic debris on sea turtles in the northwestern Gulf of Mexico, in: Shomura, R.S., Godfrey, M.L. (Eds.), Proceedings of the Second International Confernece on Marine Debris. NOAA Technical Memorandum NMFS-TM-SWFSC-154.

Plotkin, P.T., 1995. National Marine Fisheries Service and U. S. Fish and Wildlife Service Status Reviews for Sea Turtles Listed under the Endangered Species Act of 1973. Silver Spring, Maryland.

Poiner, I.R., Harris, A.N.M., 1996. Incidental capture, direct mortality and delayed mortality of sea turtles in Australia's Northern Prawn Fishery. Mar. Biol. 125, 813–825.

Poloczanska, E.S., Limpus, C.J., Hays, G.C., 2009. Vulnerability of Marine Turtles to Climate Change, in: Sims, D.W. (Ed.), Advances in Marine Biology. Burlington, Academic Press, pp. 151–211.

Ponwith, B.J., 1990. Turtle observations from Rose Atoll 18 October to 28 November, 1990. Department of Marine and Wildlife Resources, American Samoa, Pago Pago, AS.

Powell, R., 1957. Breeding turtles for profit. South Pacific Comm. Q. Bull. 7, 41–42.

Prescott, N., Pole, F., Kami, V., Hoponoa, T., Ngaluafe, P., Palaki, A., Matoto, L., Samani, T., 2004. Tonga biodiversity stocktaking. Technical Report No. 1. For The Development Of A National Biodiversity Strategic Action Plan (NGSAP). Department of Environment.

Prieto, D.A., Hernández, J.L., Sánchez, L.A., Ortega, J.G., 2012. Reproductive success of the nesting population of green turtles on Aves Island, Venezuela, in: Jones, T.T., Wallace, B.P. (Eds.), Proceedings of the Thirty- First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NOAA NMFS-SEFSC-631, San Diego, California, pp. 59–60.

Primack, 1998. Essentials of Conservation, Second edi. ed. Sinauer associates Inc., USA.

Prince, R.I.T., 1993. Western Australian marine turtle conservation project: an outline of scope and an invitation to participate. Mar. Turt. Newsl. 60, 8–14.

Prince, R.I.T., 2003. The Western Australian Marine Turtle Project, in: Ningaloo Marine Turtle Conference.

Pritchard, P.C.H., 1969. Sea turtles of the Guianas. Bull. Florida Stat. Mus. 13, 85–140.

Pritchard, P.C.H., 1971. Galapagos Sea Turtles: Preliminary Findings. J. Herpetol. 5, 1–9.

Pritchard, P.C.H., 1994. Les D'Entrecasteaux Elfin! Report of an expedition to study the sea turtles of the D'Entrecasteaux reefs, north of New Caledonia, in: Schroeder, B.A., Witherington, B.E. (Eds.), Proceedings of the Thirteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Tech. Memo. NMFS-SEFSC-341, pp. 143–145.

Pritchard, P.C.H., 1995. Marine turtles of Micronesia, in: Biology and Conservation of Sea Turtles. Washington, DC: Smithsonian Institution Press, pp. 263–274.

Pritchard, P.C.H., 1997. Evolution, Phylogeny, and current status. CRC Press, Boca Raton, pp. 1–28.

Pritchard, P.C.H., Trebbau, P., 1984. The turtles of Venezuela. Society for the Study of Amphibians and Reptiles.

Prochazka, K., Davies, B., Griffiths, C., Hara, M., Luyeye, N., O'Toole, M., Bodenstein, J., Probyn, T., Clark, B., Earle, A., Tapscott, C., Hasler, R., 2005. Benguela Current, GIWA Regional assessment 44. UNEP. Univ. Kalmar, Kalmar, Sweden. 92 pp.

Proietti, M.C., Lara-Ruiz, P., Reisser, J.W., Pinto, L. da S., Dellagostin, O.A., Marins, L.F., 2009. Green turtles (*Chelonia mydas*) foraging at Arvoredo Island in Southern Brazil: genetic characterization and mixed stock analysis through mtDNA control region haplotypes. Genet. Mol. Biol. 32.

Proietti, M.C., Reisser, J.W., Kinas, P.G., Kerr, R., Monteiro, D.S., Marins, L.F., Secchi, E.R., 2012. Green turtle *Chelonia mydas* mixed stocks in the western South Atlantic, as revealed by mtDNA haplotypes and drifter trajectories. Mar. Ecol. Prog. Ser. 447, 195–209.

Project GloBAL, 2007. Regional assessment: Western Indian Ocean. Project GloBal - Global Bycatch Assessment of Long-Lived Species Report. 14 pages.

Project GloBAL, 2009a. Country profile: Guam. Project GloBal - Global Bycatch Assessment of Long-Lived Species Report. 19 pages.

Project GloBAL, 2009b. Country profile: Northern Marianas Islands. Project GloBal - Global Bycatch Assessment of Long-Lived Species Report. 15 pages.

Project GloBAL, 2009c. Country profile: Nauru. Project GloBal - Global Bycatch Assessment of Long-Lived Species Report. 13 pages.

Project GloBAL, 2009d. Country profile: Papua New Guinea. Project GloBal - Global Bycatch Assessment of Long-Lived Species Report. 23 pages.

Project GloBAL, 2009e. Country profile: Solomon Islands. Project GloBal - Global Bycatch Assessment of Long-Lived Species Report. 12 pages.

Project GloBAL, 2009f. Country profile: Tokelau. Project GloBal - Global Bycatch Assessment of Long-Lived Species Report. 13 pages.

Projecto Tamar, 2011. Projeto TAMAR Database (SITAMAR).

Prosdocimi, L., González Carman, V., Albareda, D.A., Remis, M.I., 2012. Genetic composition of green turtle feeding grounds in coastal waters of Argentina based on mitochondrial DNA. J. Exp. Mar. Bio. Ecol. 412, 37–45.

Pulea, M., 1992. Legislative Review of environmental Law, Cook Islands. SPREP Regional Tech Assistance Project II Title III (Series).

Pultz, S., Krueger, S., McSharry, H., Balazs, G.H., 1999. Marine turtle survey on Tinian, Mariana Islands. Micronesica 32, 85–94.

Quirós Pereira, W., Figgener, C., 2011. Proyecto de conservacion e investigacioin de la tortuga baula y negra: Refugio Nacional de Vida Silvestres Ostional Informe Temporada 2011/2012. Unpublished Internal Report MINAET.

Rabalais, S.C., Rabalais, N.N., 1980. The occurrence of sea turtles on the south Texas coast. Contrib. Mar. Sci. 23, 123–129.

Rader, H.A., Ela Mba, M.A.A., Morra, W., Hearn, G.W., 2006. Marine turtles on the southern coast of Bioko Island (Gulf of Guinea, Africa), 2001–2005. Mar. Turt. Newsl. 8–10.

Raidal, S.R., Prince, R.I.T., 1996. First confirmation of multiple fibropapillomas in a Western Australian green turtle (*Chelonia mydas*). Mar. Turt. Newsl. 74, 7–9.

Rainey, W.E., 1971. Reconnaissance of the green turtle, *Chelonia mydas*, nesting aggregation at Aves Island, Lesser Antilles. Caribbean Res. Inst., College of Virgin Islands, St. Thomas, V.I.

Rajakaruna, R.S., Naveen J. Dissanayake, D.M., Lalith Ekanayake, E.M., Ranawana, K.B., 2009. Sea turtle conservation in Sri Lanka: assessment of knowledge, attitude and prevalence of consumptive use of turtle products among coastal communities. Indian Ocean Turt. Newsl. 10, 1–13.

Rakotonirina, B., 1987. Les tortues marines dans le sud de Madagascar. Etude bibliographique et enquetes aupres des pecheurs. Mem. DEA d'Oceanologie Appl. Univ. Toliara 70.

Rakotonirina, B., Cooke, A., 1994. Sea turtles of Madagascar–their status, exploitation and conservation. Oryx 28, 51–61.

Ramirez-de Veyra, R.T.D., 1994. Status of marine turtles in the Philippines, in: Bjorndal, K.A., Bolten, A.B., Johnson, D.A., Eliazar, P.J. (Eds.), Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-351, Hilton Head, South Carolina, pp. 123–125.

Rashid, S.M.A., 2006. Marine turtle conservation activities in Southeastern coastal ecologically critical areas of Bangladesh, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Proceedings of the Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece, p. 116.

Read, T.C., Fonfreyde, C., 2011. Les récifs d'Entrecasteaux Synthèse des missions de suivi terrestre des tortues marines. Service de al Marine Marchande et des Peches Maritimes.

Reardon, R., Mansfield, K.L., 1997. Annual report-1997 season: Dry Tortugas National Park sea turtle monitoring program, Monroe County, Florida. National Park Service unpublished report.

Rebelo, R., Barbosa, C., Granadeiro, J.P., Indjai, B., Novais, B., Rosa, G.M., Catry, P., 2011. Can leftovers from predators be reliably used to monitor marine turtle hatchling sex-ratios? The implications of prey selection by ghost crabs. Mar. Biol. 159, 613–620.

Redfoot, W.E., 1997. Population structure and feeding ecology of green turtles utilizing the Trident Submarine Basin, Cape Canaveral, Florida as developmental habitat. Univ. of Central Florida, Orlando, FL.

Redfoot, W.E., Ehrhart, L.M., 2000. Green turtles in three developmental habitats of the Florida Atlantic coast: population structure, fibropapillomatosis and post-juvenile migratory destinations, in: Abreu-Grobois, F.A., Briseno-Duenas, R., Marquez, R., Sarti, L. (Eds.), Proceedings of the Eighteenth International Sea Turtle Symposium. U.S. Dept. of Commerce. NOAA Technical Memorandum NMFS-SEFSC-436, p. 32.

Redfoot, W.E., Ehrhart, L.M., 2013. Trends in size class distribution, recaptures, and abundance of juvenile green turtles (*Chelonia mydas*) utilizing a rock riprap lined embayment at Port Canaveral, Florida, USA, as developmental habitat. Chelonia Conserv. Biol. 12, 252–261.

Redlow, T., Foley, A., Singel, K., 2002. Sea turtle mortality associated with red tide events in Florida, in: Seminoff, J.A. (Ed.), Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503, p. 272.

Rees, A.F., Al-Kiyumi, A., Broderick, A.C., Papathanasopoulou, N., Godley, B.J., 2012. Each to Their Own: Inter-Specific Differences in Migrations of Masirah Island Turtles. Chelonia Conserv. Biol. 11, 243–248.

Rees, A.F., Hafez, A. Al, Lloyd, J.R., Papathanasopoulou, N., Godley, B.J., 2013. Green turtles, *Chelonia mydas*, in Kuwait: nesting and movements. Chelonia Conserv. Biol. 12, 157–163.

Rees, A.F., Saad, A., Jony, M., 2008. Discovery of a regionally important green turtle *Chelonia mydas* rookery in Syria. Oryx 42, 456–459.

Rees, A.F., Saad, A., Jony, M., 2010. Syria, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Policies. IUCN, Gland, Switzerland, pp. 233–243.

Regional Organization for the Conservation of the Environment of the Red Sea and Gulf of Aden (PERSGA), 2001. Strategic action programme for the Red Sea and Gulf of Aden: country reports.

Reich, K.J., Bjorndal, K.A., Bolten, A.B., 2007. The “lost years” of green turtles: using stable isotopes to study cryptic lifestages. Biol. Lett. 3, 712–4.

Reich, K.J., Bjorndal, K.A., Martinez del Rio, C., 2008. Effects of growth and tissue type on the kinetics of 13C and 15N incorporation in a rapidly growing ectotherm. Oecologia 155, 651–663.

Reichart, H.A., Fretey, J., 1993. WIDECAST Sea Turtle Recoverv Action Plan for Suriname, in: Eckert, K.L. (Ed.), CEP Technical Report No. 24. UNEP Caribbean Environment Programme, Kingston, Jamaica, p. 65.

Reid, J.L., 2003. On the total geostrophic circulation of the Indian Ocean: flow patterns, tracers, and transports. Prog. Oceanogr. 56, 137–186.

Reischig, T., Basuki, N.R., Moord, V.A., Cordes, H., Latorra, R., 2012. Green turtles (*Chelonia mydas*) in the Berau Archipelago, Indonesia: Population assessment, nesting activities, and protection status, in: Jones, T.T., Wallace, B.P. (Eds.), Proceedings of the 31st Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-631, San Diego, California, p. 243.

Ricardo, J.A., Martin, M.E.I., Fernandez, R.D., Urquiola, A.R., Carrasco, Y.R., Castellanos, C.Y.L., Tamayo, D.R., 2006. Reproductive success of *Chelonia mydas* in nesting areas of Guanahacabibes, Cuba, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece, p. 318.

Richards, P.M., Epperly, S.P., Heppell, S.S., King, R.T., Sasso, C.R., Moncada, F.G., Nodarse, G., Shaver, D.J., Medina, Y., Zurita, J.C., 2011. Sea turtle population estimates incorporating uncertainty: a new approach applied to western North Atlantic loggerheads *Caretta caretta*. Endanger. Species Res. 15, 151–158.

Richardson, P.B., Broderick, A.C., Coyne, M.S., Ekanayake, E.M.L., Kapurusinghe, T., Premakumara, C., Ranger, S., Saman, M.M., Witt, M.J., Godley, B.J., 2013. Satellite telemetry reveals behavioural plasticity in a green turtle population nesting in Sri Lanka. Mar. Biol. 160, 1415–1426.

Rivas-Zinno, F., 2012. Captura incidental de tortugas marinas en Bajos del Solis, Uruguay. Universidad de la Republica Uruguay, Departamento de Ecologia y Evolucion.

Roberts, K.J., Ehrhart, L.M., 2007. Impacts of beach nourishment on loggerhead and green turtles in Brevard County, Florida. Biol. Lett. 3, 712–714.

Roberts, M.A., Schwartz, T.S., Karl, S.A., 2004. Global population genetic structure and male-mediated gene flow in the green sea turtle (*Chelonia mydas*): analysis of microsatellite loci. Genetics 166, 1857.

Robins, C.M., Goodspeed, A.M., Poiner, I.R., Harch, B.D., 2002. Monitoring the Catch of Turtles in the Northern Prawn Fishery. Fisheries Research and Development Corporation Final Report, Canberra. This.

Roden, S.E., Dutton, P.H., Morin, P.A.P.A., 2009. Characterization of single nucleotide polymorphism markers for the green sea turtle (*Chelonia mydas*). Mol. Ecol. Resour. 9, 1055–1060.

Roden, S.E., Morin, P.A., Frey, A., Balazs, G.H., Zárate, P., Cheng, I.-J., Dutton, P.H., 2013. Green turtle population structure in the Pacific: new insights from single nucleotide polymorphisms and microsatellites. Endanger. Species Res. 20, 227–234.

Rodrigues, H.A., Soto, J.M.R., Serafini, T.Z., Corbetta, R., 2012. Historical occurrence and characterization of cutaneous fibropapillomas in the green turtle, *Chelonia mydas*, along the Southern Brazilian Coast. Mar. Turt. Newsl. 133, 17–20.

Rögl, F., 1998. Palaeogeographic Considerations for Mediterranean and Paratethys Seaways (Oligocene to Miocene).

Roos, D., Pelletier, D., Ciccione, S., Taquet, M., Hughes, G.R., 2005. Aerial and snorkelling census techniques for estimating green turtle abundance on foraging areas: A pilot study in Mayotte Island (Indian Ocean). Aquat. Living Resour. 198, 193–198.

Ross, J.P., 1979. Sea turtles in the Sultanate of Oman. WWF Project 1320.

Ross, J.P., 1987. Sea turtle management plan for the Sultanate of Oman. Marine Science and Fisheries Center, Ministry of Agriculture and Fisheries, Sultanate of Oman, and Omani-American Joint Commission.

Ross, J.P., Barwani, M.A., 1982. Review of sea turtles in the Arabian area, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, pp. 373–383.

Ross, P., DeLorenzo, M.E., 1997. Sediment contamination problems in the Caribbean Islands: Research and regulation. Environ. Toxicol. Chem. 16, 52–58.

RPS Environmental Pty Ltd., 2008. INPEX environmental impact assessment studies - Technical appendix: Marine ecology. Prepared for INPEX Browse, Ltd. Perth.

Ruddle, K., 1996. Back to first "design principles": the issue of clearly defined boundaries. SPC Tradit. Mar. Resour. Manag. Knowl. Inf. Bull. 6, 4–12.

Rudrud, R.W., Kroeker, J.W., Leslie, H.Y., Finney, S.S., 2007. The sea turtle wars: Culture, war and sea turtles in The Republic of the Marshall Islands. SPC Tradit. Mar. Resour. Manag. Knowl. Inf. Bull. 21, 3–29.

Ruggieri, G.., Adams, C.J., Ager, D.V., 1967. The Miocene and latter evolution of the Mediterranean Sea. Aspects of Tethyan Biogeography. Systematic Association Publication., London, England.

Ruiz, A., Diaz, M., Merel, R., 2007. WIDECAST Plan de Accion para la Recuperacion de las Tortugas Marinas de Panama, in: Guada, H.J. (Ed.), Informe Tecnico Del PAC No. 47. Caribbean Environment Programme, Kingston, p. 119.

Ruiz-Urquiola, A., Riverón-Giró, F.B.F., Pérez-Bermúdez, E., Abreu-Grobois, F.A., González-Pumariega, M., James-Petric, B., Díaz-Fernández, R., Álvarez-Castro, J., Jager, M., Azanza Ricardo, J., Espinosa-López, G., González-Pumariega, Maribel James-Petric, B.L., Álvarez-Castro, J.M., 2010. Population genetic structure of greater Caribbean green turtles (*Chelonia mydas*) based on mitochondrial DNA sequences, with an emphasis on rookeries from southwestern Cuba. Rev. Investig. Mar. 31, 33–52.

Rumbold, D.G., Davis, P.W., Perretta, C., 2001. Estimating the effect of beach nourishment on *Caretta caretta* (loggerhead sea turtle) nesting. Restor. Ecol. 9, 304–310.

Rupeni, E., Mangubhai, S., Tabunakawai, K., Blumel, P., 2002. Establishing replicable community-based turtle conservation reserves in Fiji, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop, Honolulu, HI, USA. pp. 119–124.

Russell, D.J., Balazs, G.H., 2009. Dietary Shifts by Green Turtles (*Chelonia mydas*) in the Kāne‘ohe Bay Region of the Hawaiian Islands: A 28-Year Study. Pacific Sci. 63, 181–192.

Saad, M.A., 1999. Hadramaut coast importance in conservation of endangered green turtle. Marine Sciences Resources Research Center, Aden. Unpublished Report. 8 pages.

Sachet, M., 1954. A summary of information on Rose Atoll. Atoll Res. Bull. 29, 1–25.

Saili, K., 2005. Investigations into the status of marine turtles in American Samoa. NOAA/NMFS Unallied Management Grant: Award No. NA03NMF4540355 FINAL REPORT (September 1st, 2003 - August 31st, 2005). In: D. o. M. a. W. Resources (ed.). DMWR, Pago Pago, American Samoa.

Sakai, H., Saeki, K., Ichihashi, H., Kamezaki, N.I., Tanabe, S., Tatsukawa, R., 2000. Growth-related changes in heavy metal accumulation in green turtle (*Chelonia mydas*) from Yaeyama Islands, Okinawa, Japan. Arch. Environ. Contam. Toxicol. 39, 378–385.

Sako, T., Horikoshi, K., 2003. Marine debris ingested by green turtles in the Ogasawara Islands, Japan, in: Seminoff, J.A. (Ed.), Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503, p. 305.

Salano Cordero, R., Fallas, G., Madrigal Castro, M.V., Rojas Malespin, J., 2010. Informe final de investigacion 2010. Evaluacion y seguimiento de la dinamica de anidacion de tortugas marinas en Playa Buena Vista, Nicoya, Costa Rica. Unpublished Report.

Salm, R.V., 1984. Sea turtle trade in Indonesia. IUCN/WWF Proj. 3108 F. Rep. #5. Mar. Conserv. Bogor, Indones. IUCN/WWF 50.

Salm, R.V., 1991. Turtles in Oman: status, threats and managment options.

Salm, R.V., Jensen, R.A.C., Papastavrou, V.A., 1993. Marine Fauna of Oman: Cetaceans, Turtles, Seabirds and Shallow Water Corals. Gland, Switzerlan.

Salm, R.V., Salm, S., 2001. Sea turtles in the Sultanate of Oman, 2nd editio. ed. The Historical Association of Oman. Sultanate of Oman. 48 p.

Salmon, M., Wyneken, J., Fritz, E., Lucas, M., 1992. Seafinding by hatchling sea turtles: role of brightness , silhouette and beach slope as orientation cues. Behaviour 122, 56–77.

Sands, N.J., Rawson, K., Currens, K.P., Graeber, W.H., Ruckelshaus, M.H., Fuerstenberg, R.R., Scott, J.B., 2009. Determination of independent populations and viability criteria for the Hood Canal summer chum salmon evolutionarily significant unit. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-101.

Santos, R.G., Martins, A.S., Farias, J.D.N., Horta, P.A., Pinheiro, H.T., Torezani, E., Baptistotte, C., Seminoff, J.A., Balazs, G.H., Work, T.M., 2011. Coastal habitat degradation and green sea turtle diets in Southeastern Brazil. Mar. Pollut. Bull. 62, 1297–302.

Sargent, F.J., Leary, T.J., Crewz, D.W., Kruer, C.R., 1995. Scarring of Florida's seagrasses: assessment and management options. Florida Department of Environmental Protection FMRI Technical Report TR-1. Florida Marine Research Institute, St. Petersburg, Florida. 37 p. plus appendices.

Sarkar, S., 2001. Threat to survival and conservation efforts in Bangladesh, in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-528, pp. 307–308.

Sasso, C.R., Epperly, S.P., 2006. Seasonal sea turtle mortality risk from forced submergence in bottom trawls. Fish. Res. 81, 86–88.

Scavia, D., Field, J.C., Boesch, D.F., Buddemeier, R.W., Burkett, V., Cayan, D.R., Fogarty, M., Harwell, M.A., Howarth, R.W., Mason, C., Denise, J., Royer, T.C., Sallenger, A.H., Titus, J.G., 2002. Climate Change Impacts on U . S . Coastal and Marine Ecosystems. Estuaries 25, 149–164.

Schmelz, G.W., Mezich, R.R., 1988. A preliminary investigation of the potential impact of Australian pines on the nesting activities of the loggerhead turtle, in: Schroeder, B.A. (Ed.), Proceedings of the Eighth Annual Workshop on Sea Turtle Conservation and Biology. NOAA Technical Memorandum NMFS-SEFSC-214, pp. 63–66.

Schoch, B.A., Wetterer, J.K., Olsen, J., 2006. Relocation of green sea turtle geggs to Bermuda, in: Pilcher, N.J. (Ed.), Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-536, p. 94.

Schroeder, B.A., Ehrhart, L.M., Bagley, D.A., Coyne, M.S., Foley, A., Balazs, G.H., Witherington, B.E., 2008. Migratory routes and resident areas of adult female and male Florida green turtles., in: Rees, A.F., Frick, M., Panagopoulou, A., Williams, K. (Eds.), Proceedings of the 27th Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-569, pp. 59–60.

Schroeder, B.A., Foley, A.M., Witherington, B.E., Mosier, A.E., 1998. Ecology of marine turtles in Florida Bay: population structure, distribution, and occurrence of fibropapilloma, in: Epperly, S.P., Braun, J. (Eds.), Proceedings of the Seventeenth Annual Sea Turtle Symposium. NOAA Technical Memorandum NMFS-SEFSC-415, pp. 265–267.

Schroeder, B.A., Mosier, A.E., 2000. Between a rock and a hard place: coastal armoring and marine turtle nesting habitat in Florida, in: Abreu-Grobois, F.A., Briseño-Dueñas, R., Marquez, R., Sarti, L. (Eds.), Proceedings of the Eighteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-436, pp. 290–292.

Schroeder, B.A., Owens, D.W., 1994. Sex ratio of immature green turtles in an east central Florida developmental habitat, in: Schroeder, B.A., Witherington, B.E. (Eds.), Proceedings of the Thirteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-341, Jekyll Island, Georgia, pp. 157–161.

Schulz, J.P., 1975. Sea turtles nesting in Surinam. Zool. Verh. 143, 1–141.

Schulz, J.P., 1982. Status of sea turtle populations nesting in Surinam with notes on Sea turtles nesting in Guyana and French Guiana, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution, pp. 435–437.

Schulz, J.P., 1984. Turtle conservation strategy in Indoensia. IUCN/WWF Report No 6.

Schulz, J.P., 1987. USE Status of and trade in *Chelonia mydas* and Eretmochelys inbricata in Indonesia. Consultancy report prepared for IUCN Conservation Monitoring Centre.

Sea Turtle Conservancy, 2013. Annual Reports [WWW Document]. URL http://www.conserveturtles.org/costarica.php?page=season-reports

Sea turtles education booklet, Viet Nam, 2012. . Thuy San, IUCN, WWF, Traffic, Sea Turtle Survial League. Accessed in 2012 http://www.ioseaturtles.org/Education/seaturtlebooklet.pdf.

Searle, L.A.W., 2003. Diet of green turtles (*Chelonia mydas*) captured in the Robinson Point foraging ground, Belize, in: Seminoff, J.A. (Ed.), Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503, pp. 228–229.

Searle, L.A.W., 2004. A brief history of sea turtle communities, conservation and consumption in Belize, in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-528, pp. 316–318.

Searle, L.A.W., 2006. The effect on new fishery regulations on traditional turtle fisherman in Belize, in: Pilcher, N.J. (Ed.), Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-536, p. 94.

Secretariat of the Pacific Community, 1997. Coastal sedimentation erosion and management of Majuro Atoll, Republic of Marshall Islands. Suva, Fiji.

Secretariat of the Pacific Community and Bureau of Marine Resources Palau, 2007. Palau domestic fishing laws.

Sella, I., 1995. Sea turtles in the Eastern Mediterranean and North Red Sea, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C., pp. 417–423.

Seminoff, J.A., 1994. Conservation of the marine turtles of Mexico, a survey of nesting beach conservation projects. Masters Thesis. Department of Ecology and Evolutionary Biology, The University of Arizona.

Seminoff, J.A., 2000. Biology of the east pacific green turtle, *Chelonia mydas* agassizii at a warm temperate feeding area in the gulf of California, Mexico. Ph.D. Thesis. University of Arizona.

Seminoff, J.A., Alfaro-Shigueto, J., Amorocho, D.F., Arauz-Vargas, R., Baquero Gallegos, A., Chacón-Chaverri, D., Gaos, A.R., Kelez, S., Mangel, J.C., Urteaga, J., Wallace, B.P., 2012a. Biology and conservation of sea turtles in the eastern Pacific Ocean, in: Seminoff, J.A., Wallace, B.P. (Eds.), Sea Turtles of the Eastern Pacific: Advances in Research and Conservation. University of Arizona Press, Tucson, Arizona, pp. 11–38.

Seminoff, J.A., Benson, S.R., Arthur, K.E., Eguchi, T., Dutton, P.H., Tapilatu, R.F., Popp, B.N., 2012b. Stable Isotope Tracking of Endangered Sea Turtles: Validation with Satellite Telemetry and $\delta 15N$ Analysis of Amino Acids. PLoS One 7, e37403.

Seminoff, J.A., Crouse, D.T., Pilcher, N.J., Eckert, K.L., Marcovaldi, M.Â., Mortimer, J.A., Nichols, W.J., 2004. 2004 global status assessment - Green turtle (*Chelonia mydas*), IUCN.

Seminoff, J.A., Jones, T.T., Resendiz, A., Nichols, W.J., Chaloupka, M.Y., 2003. Monitoring green turtles (*Chelonia mydas*) at a coastal foraging area in Baja California, Mexico: multiple indices to describe population status. J. Mar. Biol. Assoc. United Kingdom 83, 1355–1362.

Seminoff, J.A., Resendiz, A., Nichols, W.J., 2002a. Home range of green turtles *Chelonia mydas* at a coastal foraging area in the Gulf of California, Mexico. Mar. Ecol. Prog. Ser. 242, 253–265.

Seminoff, J.A., Resendiz, A., Nichols, W.J., 2002b. Diet of East Pacific Green Turtles (*Chelonia mydas*) in the Central Gulf of California, México. J. Herpetol. 36, 447–453.

Seminoff, J.A., Resendiz, A., Nichols, W.J., Jones, T.T., 2002c. Growth Rates of Wild Green Turtles (*Chelonia mydas*) at a Temperate Foraging Area in the Gulf of California, México. Copeia 610–617.

Seminoff, J.A., Schroeder, B.A., MacPherson, S.L., Possardt, E.E., Bibb, K., 2007. Green sea turtle (*Chelonia mydas*) 5-year review: summary and evaluation. NMFS and USFWS, Silver Spring, Maryland, and Jacksonville, Florida.

Seminoff, J.A., Zárate, P., Coyne, M.S., Foley, D.G.D., Parker, D.M., Lyon, B.N., Dutton, P.H., 2008. Post-nesting migrations of Galápagos green turtles *Chelonia mydas* in relation to oceanographic conditions: integrating satellite telemetry with remotely sensed ocean data. Endanger. Species Res. 4, 57–72.

Senol, A., 1999. Altınkum and North Karpas Beaches Marine Turtles Research, Research Report.

Senol, A., 2001. South and North Karpas Beaches Marine Turtles Research, Research Report.

Senol, A., Kusetogulları, H., 2002. Monitoring and protection of sea turtles, project report 2002.

Sereywath, P., 2006. Status of sea turtle conservation program in Cambodia, in: Pilcher, N.J. (Ed.), Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-536, p. 261.

Serra, G., 2008. Preliminary report on natural heritage sites in western Syria.

Settle, S., 1995. Status of nesting populations of sea turtles in Thailand and their conservation. Mar. Turt. Newsl. 68, 8–13.

Shaffer, M., 1987. Minimum viable populations: coping with uncertainty, in: Soule, M.E. (Ed.), Viable Populations for Conservation. Cambridge University Press, pp. 69–86.

Shamblin, B.M., Bjorndal, K.A., Bolten, A.B., Hillis-Starr, Z.M., Lundgren, I., Naro-Maciel, E., Nairn, C.J., 2012a. Mitogenomic sequences better resolve stock structure of southern Greater Caribbean green turtle rookeries. Mol. Ecol.

Shamblin, B.M., Bolten, A.B., Bjorndal, K.A., Dutton, P.H., Nielsen, J.T., Abreu-Grobois, F.A., Reich, K.J., Witherington, B.E., Bagley, D.A., Ehrhart, L.M., Tucker, A.D., Addison, D.S., Arenas, A., Johnson, C., Carthy, R.R., Lamont, M.M., Dodd, M.G., Gaines, M.S., LaCasella, E.L., Nairn, C.J., 2012b. Expanded mitochondrial control region sequences increase resolution of stock structure among North Atlantic loggerhead turtle rookeries. Mar. Ecol. Prog. Ser. 469, 145–160.

Shanker, K., 2004. Marine turtle status and conservation in the Indian Ocean, in: Expert Consultation on Interactions between Sea Turtles and Fisheries within an Ecosystem Context. FAO Fisheries Report No. 738, Rome, Italy, pp. 85–134.

Shanker, K., Andrews, H. V., 2004. Tracing the migration of Indian marine turtles: Towards an integrated and collaborative conservation program. Interim Report-June 2004, Centre for Herpetology/Madras Crocodile Bank Trust. Tamil Nadu.

Shanker, K., Pilcher, N.J., 2003. Marine Turtle Conservation in South and Southeast Asia: Hopeless Cause or Cause for Hope? Mar. Turt. Newsl. 100, 43–51.

Shapiro, S., Bruckner, S., Lau, J., Reilly, J., Zacharski, M., Zhang, Y., 2013. Analysis of the federal oil and gas regulatory landscape. A Rep. Prep. Occup. Saf. Heal. Adm. 69 pages.

Shaver, D.J., 1990. Hypothermic stunning of sea turtles. Mar. Turt. Newsl. 48, 25–27.

Shaver, D.J., 1994. Relative abundance, temporal patterns, and growth of sea turtles at the Mansfield Channel, Texas. J. Herpetol. 28, 491–497.

Shaver, D.J., 2002. Green sea turtles (*Chelonia mydas*) in a south Texas (USA) developmental habitat, in: Mosier, A., Foley, A., Brost, B. (Eds.), Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-477, p. 9.

Shaver, D.J., Plotkin, P.T., 1998. Marine debris ingestion by sea turtles in South Texas: Pre- and post-MARPOL ANNEX V, in: Byles, R., Fernandez, Y. (Eds.), Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS- SEFSC-412, Hilton Head, South Carolina, p. 124.

Simms, C.N., Clarke, M., Campbell, A.C., 2002. Ghost crab predation of loggerhead turtle hatchlings in the Sinai Region of Egypt, in: Mosier, A., Foley, A., Brost, B. (Eds.), Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFC-477, pp. 202–203.

Singel, K., Redlow, T., Foley, A., 2003. Twenty-two years of data on sea turtle mortality in Florida: trends and factors, in: Seminoff, J.A. (Ed.), Proceedings of the Twenty-Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503, p. 275.

Siow, K.T., Moll, E.O., 1982. Status and conservation of estuarine and sea turtles in West Malaysian waters, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, pp. 339–347.

Smith, G.W., 1992. Hawksbill Turtle Nesting at Manatee Bar, Belize, 1991. Mar. Turt. Newsl. 57, 1–5.

Smith, G.W., Eckert, K.L., Gibson, J.P., 1992. WIDECAST Sea Turtle Recovery Action Plan for Belize (Karen L. Eckert, Editor). CEP Tech. Rep. No. 18 UNEP Caribb. Environ. Program. Kingston, Jamaica 86 pages.

Smith, M.M., Salmon, M., 2009. A comparison between the habitat choices made by hatchling and juvenile green turtles (*Chelonia mydas*) and loggerheads (*Caretta caretta*) 9–13.

Smith, R., Edward, A., Shorten, G., 1997. Sand dredging, Nett Municipality Pohnpei Lagoon, Federated States of Micronesia: resource assessment and environmental considerations. SOPAS Technical Report 244, April 1997.

Snoddy, J.E., Southwood Williard, A., 2010. Movements and post-release mortality of juvenile sea turtles released from gillnets in the lower Cape Fear River, North Carolina, USA. Endanger. Species Res. 12, 235–247.

Snover, M.L., Heppell, S.S., 2009. Application of diffusion approximation for risk assessments of sea turtle populations. Ecol. Appl. 19, 774–785.

Solano Martinez, M.., 2013. Convencion Interamericana para la Proteccion y Conservacion de las Tortugas Marinas, Costa Rica. Informe Annual 2013. SINAC, MINAE, San Jose Costa Rica.

Sole, G., Azara, C.E., 1996. Fibropapillomas in the green turtle (*Chelonia mydas*) of Aves Island., in: Proceedings of the 16th Annual Symposium of Sea Turtle Biology and Conservation. U.S. Department of Commerce, NOAA Technical Memo, p. 128.

Sole, G., Medina, G., 1989. The green turtles of Aves Island, in: Eckert, S.A., Eckert, K.L., Richardson, T.H. (Eds.), Proceedings of the Ninth Annual Workshop on Sea Turtle Conservation and Biology. NOAA Technical Memorandum NMFS-SEFC-23, Jekyll Island, Georgia, pp. 171–173.

Solomon Islands Ministry of Environment Conservation and Meteorology, 2008. Solomon Islands state of the environment report 2008. Prepared by Pacific Horizon Consultancy Group (Honiara, Solomon Islands) for Ministry of Environment, Conservation and Meteorology (Honiara, Solomon Islands). 97 pages.

Solomon, J.L., Blumenthal, J.M., 2008. Cayman Islands Government, Department of Environment Annual Marine Turtle Beach Monitoring Program 2008. Unpublished report.

Solomon, J.L., Blumenthal, J.M., 2009. Cayman Islands Government, Department of Environment Annual Marine Turtle Beach Monitoring Program 2007. Unpublished report.

Solomon, J.L., Blumenthal, J.M., Austin, T.J., Ebanks-Petrie, G., Broderick, A.C., Godley, B.J., 2006. Insights into the nesting population of marine turtles in the Cayman Islands, in: Frick, M.A., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Twenty-Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece, Athens, Greece, pp. 323–324.

Solow, A.R., Bjorndal, K.A., Bolten, A.B., 2002. Annual variation in nesting numbers of marine turtles: the effect of sea surface temperature on re-migration intervals. Ecol. Lett. 5, 742–746.

Spalding, M.D., Fox, H.E., Allen, G.R., Davidson, N., Ferdana, Z.A., Finlayson, M., Halpern, B.S., Jorge, M.A., Lombana, A., Lourie, S.A., Martin, K.D., McManus, E., Molnar, J., Recchia, C.A., Robertson, J., 2007. Marine ecoregions of the world: a bioregionalization of coastal and shelf areas. Bioscience 57, 573–583.

Spotila, J.R.R., Standora, E.A.A., Morreale, S.J.J., Ruiz, G.J.J., 1987. Temperature dependent sex determination in the green turtle (*Chelonia mydas*): effects on the sex ratio on a natural nesting beach. Herpetologica 43, 74–81.

SPREP, 2012. Pacific Islands regional marine species programme 2012-2017. Apia, Samoa.

Stancyk, S.E., 1982. Non-human predators of sea turtles and their control, in: Bjorndal, K.A. (Ed.), Biology and Conservation of Sea Turtles. Smithsonian Institution Press, Washington D.C., pp. 139–152.

Stanton, D.B., 2008. Ras Sharma Protected Area remains unprotected. Wildl. Middle East News 3, 7.

Steinitz, M.J., Salmon, M., Wyneken, J., 1998. Beach renourishment and loggerhead turtle reproduction: a seven year study at Jupiter Island, Florida. J. Coast. Res. 14, 1000–1013.

Sterling, E.J., McFadden, K.W., Holmes, K.E., Vintinner, E.C., Arengo, F., Naro-Maciel, E., 2013. Ecology and Conservation of Marine Turtles in a Central Pacific Foraging Ground.

Stewart, K.R., Wyneken, J., 2004. Predation risk to loggerhead hatchlings at a high-density nesting beach in southeast Florida. Bull. Mar. Sci. 74, 325–335.

Stiles, D., 2009. The marine turtle product trade in Viet Nam. Mar. Turt. Newsl. 6–9.

Still, B., Tuxbury, K., Prescott, R., Ryder, C., Murley, D., Merigo, C., Smith, C., Turnbull, B., 2002. A record cold stun season in Cape Cod Bay, Massachussets, USA, in: Mosier, A., Foley, A., Brost, B.. (Eds.), Proceedings of the Twentieth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-477, pp. 205–206.

Stoddart, D.R., Gibbs, P.E., Hopley, D., 1981. Natural history of Raine Island, Great Barrier Reef. Atoll Res. Bull. 254, 1–137.

Stokes, L., Stokes, D., Grimanis, K., Pantis, J.D., 2011. Episkopi Bay, Southern Cyprus: increased number of strandings of dead *Chelonia mydas* reveal an important foraging ground for green turtles in the Mediterranean, in: Book of Abstracts. 4th Mediterranean Conference on Marine Turtles Naples - Italy. p. 66.

Suganuma, H., 1985. Green turtle research program in Ogasawara. Mar. Turt. Newsl. 33, 2–3.

Suganuma, H., Horikoshi, K., Tachikawa, H., Sato, F., Yamaguchi, M., 1996. Reproductive characteristics of the Ogasawara green turtles, in: Keinath, J.A., Barnard, D.E., Musick, J.A., Bell, B.A. (Eds.), Fifteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-387, p. 318.

Sulu, R., Hay, C., Ramohia, P., Lam, M., 2000. The status of Solomon Islands coral reefs.

Sunderraj, S.F.W., Joshua, J., Brahmbhatt, L., Saravanakumar, A., Muthuraman, B., Das, S.K., 2006a. The status of sea turtle populations on the Gujarat coast, in: Shanker, K., Andrews, H. V. (Eds.), Towards an Integrated and Collaborative Sea Turtle Conservation Programme in India: A UNEP/CMS-IOSEA Project Report. Centre for Herpetology/Madras Crocodile Bank Trust, Post Bag 4, Mamallapuram, Tamil Nadu.

Sunderraj, S.F.W., Joshua, J., Kumar, V.V., 2006b. Sea turtles and their nesting habitats in Gujarat, in: Shanker, K., Choudhury, B.C. (Eds.), Marine Turtles of the Indian Subcontinent. Universities Press, Hyderabad, Hyderabad, India, pp. 156–169.

Sunderraj, S.F.W., Joshua, J., Serebiah, S., 2001. Sea Turtles along the Gujarat Coast. Kachhapa #5 14–16.

Surjadi, P., Anwar, K., 2001. Exploitation of marine resources in the Togean and Banggai Islands, Sulawesi, Indonesia, in: Allen, G.R., McKenna, S.A. (Eds.), A Marine Rapid Assessment of the Togean and Banggai Islands, Sulawesi, Indonesia. RAP Bulletin of Biological Assessment 20, Conservation International, Washington D.C., p. 59.

Suwelo, I.S., 1971. Paper no. 14. Sea turtles in Indonesia, in: Marine Turtles. IUCN Publications New Series, 31, Morges, Switzerland, pp. 85–89.

Swimmer, Y., 2006. Relationship Between Basking and Fibropapillomatosis in Captive Green Turtles (*Chelonia mydas*). Chelonian Conserv. Biol. 5, 305–309.

SWOT, 2011. State of the World's Sea Turtles Report, Vol. VI.

Tagarino, A., 2011. Investigations into the status of marine turtles of Ofu and Olosega, American Samoa: An intensive monitoring of hawksbill nesting beaches FY ‘11-‘13, semi-annual report #2, grant number: NA10NMF4540387. Department of Marine and Wildlife Resources, American Samoa Government, Pago Pago, American Samoa.

Tagarino, A., 2012. Investigations into the status of marine turtles of Ofu and Olosega, American Samoa: an intensive monitoring of hawksbill nesting beaches FY ‘11-‘13, semi-annual report #4, grant number: NA10NMF4540387. In: D. o. M. a. W. Resources (ed.). Department of Marine and Wildlife Resources, Pago Pago, American Samoa.

Tagarino, A., Saili, K., Utzurrum, R., 2008. Investigations into the status of marine turtles in American Samoa with remediation of identified threats and impediments to conservation and recovery of species. NOAA/NMFS unallied management grant: Award No. NA04NMF4540126 Final Report (01 October 2004). Department of Marine and Wildlife Resources, American Samoa Government, Pago Pago, American Samoa.

Tagarino, A., Utzurrum, R., 2010. Investigations into the status of marine turtles in American Samoa: Assessment of threat to nesting activities and habitat in Swains Island, NOAA/NMFS Unallied Management Grant: Award No. NA08NMF4540506, final report (01 October 2008 to 30 September 2009). Department of Marine and Wildlife Resources, American Samoa Government, Pago Pago, American Samoa.

Tan, P.J.B., 2004. Island eco-tourism-A case study of Penghu Archipelago, in: Islands of the World VIII International Conference. 1-7 November 2004. “Changing Islands-Changing Worlds,” Kinmen Island, Taiwan.

Tanner, C., 2013. Sea turtle conservation in Ghana’s western region: the bigger picture. Mar. Turt. Newsl. 136, 9–12.

Taquet, C., Taquet, M., Dempster, T., Soria, M., Ciccione, S., Roos, D., Dagorn, L., 2006. Foraging of the green sea turtle *Chelonia mydas* on seagrass beds at Mayotte Island (Indian Ocean), determined by acoustic transmitters. Mar. Ecol. Prog. Ser. 306, 295–302.

Tayab, M.R., Quiton, P., 2003. Marine turtle conservation initiatives at Ras laffan Industrial City, Qatar (Arabian Gulf). Mar. Turt. Newsl. 14–15.

Taylor, B.L., Dizon, A.E., 1999. First policy then science: why a management unit based solely on genetic criteria cannot work. Mol. Ecol. 8, S11–S16.

Te Honu Tea, 2007. Projet Tikehau. Project report on the study of 2007-2008 sea turtle nesting season onf the island of Tikehau. Unpublished (in French).

Teas, W.G., 1994. 1993 Annual Report of the Sea Turtle Stranding and Salvage Network: Atlantic and Gulf Coasts of the United States January-December 1993.

Teas, W.G., Witzell, W.N., 1996. Effects of anthropogenic debris on marine turtles in the western North Atlantic Ocean, in: Keinath, J.A., Barnard, D.E., Musick, J.A., Bell, B.A. (Eds.), Proceedings of the Fifteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-387, p. 323.

Teclemariam, Y., Giotom, M., Mengstu, T., Abraha, H., Mahmud, S., 2009. An update on marine turtles in Eritrea, Red Sea. Indian Ocean Turt. Newsl. 6–10.

Teeb'aki, K., 1992. Republic of Kiribati. Unpublished.

Thomas, P.E.J. (complier), 1989. Report of the Northern Marshall Islands natural diversity and protected areas survey 7-24 September 1988. South Pacific Regional Environment Programme (SPREP), Noumea, New Caledonia, and the East-West Center, Honolulu, Hawaii.

Thorbjarnarson, J.B., Platt, S.G., Khaing, S.T., 2000. Sea turtles in Myanmar: past and present. Mar. Turt. Newsl. 88, 10–11.

Tiburcio Pintos, G., Bravo Gamboa, P.R., Kinzel, M.R.., 2004. Satellite telemetry green sea turtles (*Chelonia mydas*) nesting in Lechugillas, Veracruz-Mexico, in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-528., pp. 275–277.

Tiburcio Pintos, G., Marquez-Aimarisa, P., del Rayo Duarte, M.A., Guzman-Poo, J.R., Calderon-Campuzano, M.F., 2007. First nesting report of black sea turtle (*Chelonia mydas* agassizi) in Baja California Sur, Mexico, in: Mast, R.B., Hutchinson, B.J., Hutchinson, A.H. (Eds.), Proceedings of the Twenty-Fourth Symposium on Sea Turtle Biology and Conservation.

Tiburcio-Pinto, G., Marquez-Almansa, P., Castillo-Leggs, V., Acevedo-Ruiz, E., Marron Fiol, R., Marron Fiol, J., Sankaran, S., 2012. Nesting report of black sea turtle (*Chelonia mydas* agassizii) in Baja California Sur, Mexico, in: Belskis, L., Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Proceedings of the Twenty-Ninth Annual Symposium on Sea Turtle Biologhy and Conservation. NOAA Tech. Memo. NOAA-NMFS-SEFSC-630, p. 33.

Tiwari, M., 2002. An evaluation of the perceived effectiveness of international instruments for sea turtle conservation. J. Int. Wildl. Law Policy 5, 145–156.

Tiwari, M., 2012. Sea Turtle surveys in the Southern Nicobar Islands: Results of surveys from February-May 1991. Indian Ocean Turt. Newsl. 16, 14–18.

Tiwari, M., Balazs, G.H., Hargrove, S., 2010. Estimating carrying capacity at the green turtle nesting beach of East Island, French Frigate Shoals. Mar. Ecol. Prog. Ser. 419, 289–294.

Tiwari, M., Bjorndal, K.A., Bolten, A.B., Bolker, B., 2006. Evaluation of density-dependent processes and green turtle *Chelonia mydas* hatchling production at Tortuguero, Costa Rica. Mar. Ecol. Prog. Ser. 326, 283–293.

Tiwol, C.M., Cabanban, A.S., 2000. All female hatchlings from the open-beach hatchery at Gulisaan Island, Turtle Islands park, Sabah. ASEAN ACADEMIC Press, Kuala Lumpur.

Tomás, J., Castroviejo, J., Raga, J.A., 1999. Sea Turtles in the South of Bioko Island (Equatorial Guinea). Mar. Turt. Newsl. 4–6.

Tomás, J., Castroviejo, J., Raga, J.A., 2000. Sea Turtles in the South of Bioko Island (Equatorial Guinea), Africa, in: Kalb, H.J., Wibbels, T. (Eds.), Proceedings of the 19th Annual Symposium on Sea Turtle Biology and Conservation. U.S. Dept. Commerce. NOAA Tech. Memo. NMFS-SEFSC-443, South Padre Island, Texas, pp. 247–250.

Tomás, J., Godley, B.J., Castroviejo, J., Raga, J. a., 2010. Bioko: critically important nesting habitat for sea turtles of West Africa. Biodivers. Conserv. 19, 2699–2714.

Tourinho, P.S., Ivar do Sul, J. a, Fillmann, G., 2010. Is marine debris ingestion still a problem for the coastal marine biota of southern Brazil? Mar. Pollut. Bull. 60, 396–401.

Trevor, A.P., 2009. Turtle Research and Monitoring Database System (TREDS) Annual Report 2009. SPREP.

Trindell, R., Arnold, D., Moody, K., Morford, B., 1998. Post-construction marine turtle nesting monitoring results on nourished beaches, in: LS Tait (Compiler), Rethinking the Role of Structures in Shore Protection. Proceedings of the 11 Th Annual National Conference on Beach Preservation Technology, St. Petersburg, Florida. pp. 77–92.

Tripathy, A., Kale, N., Shanker, K., 2012. Strengthening the network for monitoring and conservation of sea turtles in India. Project report submitted to the USFWS. Madras Crocodile Bank Trust and Dakshin Foundation, Bangalore. 58 pages.

Tripathy, B., Choudhury, B.C., Shanker, K., 2002. Marine turtles Of Lakshadweep Islands, India. Kachhapa 7, 3–7.

Tripathy, B., Shanker, K., Choudhury, B.C., 2006. Sea turtles and their habitats in the Lakshadweep Islands, in: Shanker, K., Choudhury, B.C. (Eds.), Marine Turtles of the Indian Subcontinent, Part 3: Marine Turtles of the Arabian Sea. Universities Press, Hyderabad, Hyderabad, India, pp. 119–136.

Troëng, S., 1998. Report on the 1997 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation.

Troëng, S., 2000a. Report on the 1999 green turtle program at Tortuguero, Costa Rica. Caribbean Conservation Corporation and the Ministry of Envirionment, Engery and Telecommunications of Costa rica.

Troëng, S., 2000b. Predation of green (*Chelonia mydas*) and leatherback (*Dermochelys coriacea*) turtles by jaguars (Panthera onca) at Tortuguero National Park, Costa Rica. Chelonian Conserv. Biol. 3, 751–753.

Troëng, S., Chaloupka, M.Y., 2007. Variation in adult annual survival probability and remigration intervals of sea turtles. Mar. Biol. 151, 1721–1730.

Troëng, S., Evans, D.R., Harrison, E., Limpus, C.J., 2005. Migration of green turtles *Chelonia mydas* from Tortuguero, Costa Rica. Mar. Biol. 148, 435–447.

Troëng, S., Rankin, E., 2005. Long-term conservation efforts contribute to positive green turtle *Chelonia mydas* nesting trend at Tortuguero, Costa Rica. Biol. Conserv. 121, 111–116.

Trono, R.B., 1991. Philippine marine turtle conservation program. Mar. Turt. Newsl. 53, 5–7.

Tuato'o-Bartley, N., Morrell, T.E., Craig, P., 1993. Status of sea turtles in American Samoa in 1991. Pacific Sci. 47, 213–221.

Tuck Jr., R.G., 1977. The turtles of Iran. Shekar Va Tabi'at 214, 20–25.

Tudela, S., 2004. Ecosystem effects of fishing in the Mediterranean: an analysis of the major threats of fishing gear and practices to biodiversity and marine habitats, General Fi. ed. Food and Agriculture Organization of the United Nations, Rome, Italy.

Türkozan, O., Durmuş, S.H., 2000. A feeding ground for juvenile green turtles, *Chelonia mydas*, on the western coast of Turkey. Br. Herpetol. Soc. Bull. 71, 1–5.

Türkozan, O., Kaska, Y., 2010. Turkey, in: Casale, P., Margaritoulis, D. (Eds.), Sea Turtles in the Mediterranean: Distribution, Threats and Conservation Policies. IUCN, Gland, Switzerland, pp. 257–293.

Türkozan, O., Özdilek, Ş.Y., Ergene, S., Uçar, A.H., Sönmez, B., Yilmaz, C., Kaçar, Y., Aymak, C., 2013. Strandings of loggerhead (*Caretta caretta*) and green (*Chelonia mydas*) sea turtles along the eastern Mediterranean coast of Turkey. Herpetol. J. 23, 11–15.

Türkozan, O., Yamamoto, K., Yılmaz, C., 2011. Nest site preference and hatching success of green (*Chelonia mydas*) and loggerhead (*Caretta caretta*) sea turtles at Akyatan Beach, Turkey. Chelonian Conserv. Biol. 10, 270–275.

Türkozan, O., Yilmaz, C., Oruç, A., 2007. Green turtle (*Chelonia mydas*) nesting on Akyatan Beach, Turkey, in: Frick, M., Panagopoulou, A., Rees, Al.F., Williams, K. (Eds.), Proceedings of the 27th Annual Symposium on Sea Turtle Biology and Conservation, Book of Abstracts. ISTS, Myrtle Beach, South Carolina, p. 274.

Turny, A., 2012. Green turtle nesting in Suriname. Personal communication.

Turtle Conservation Project, 2009. Threats to marine turtle populations in Sri Lanka [WWW Document]. URL www.tcpsrilanka.org

UNEP-MAP-RAC/SP, 2010. Impact of climate change on marine and coastal biodiversity in the Mediterranean Sea. Current state of knowledge. By S. Ben Haj and A. Liman, RAC/SPA Edit, Tunis:1:28.

United Nations Environment Programme, 2007. Lebanon post-conflict environmental assessment. United Nations Environment Programme, Nairobi, Kenya.

United Nations Environment Programme, 2011. Satellite trakcing of marine turtles in the Mediterranean. Current knowledge and conservation implications. UNEP(DEPI)/MED WG.359/inf.8 Rev.1.

Upadhyay, S., Upadhyay, V., 2002. International and national instruments and marine turtle conservation in India. J. Int. Wildl. Law Policy 5, 65–86.

U.S. Fish and Wildlife Service, 2012. Rose Atoll draft comprehensive conservation plan and environmental assessment. Rose Atoll National Wildlife Refuge/Marine National Monument, National Park Service, Pago Pago, American Samoa and USFWS, Honolulu, HI.

Utzurrum, R., 2002. Sea turtle conservation in American Samoa, in: Kinan, I. (Ed.), Proceedings of the Western Pacific Sea Turtle Cooperative Research and Management Workshop. February 5–8, 2002. Honolulu, HI: Western Pacific Regional Fishery Management Council, Honolulu, Hawaii, USA, pp. 33–36.

Valdéz-González, M. a., Ramírez-Bautista, A., 2002. Reproductive Characteristics of the Spiny Lizards, Sceloporus horridus and Sceloporus spinosus (Squamata: Phrynosomatidae) from México. J. Herpetol. 36, 36–43.

Vallini, C., Annibale, O., Rubini, S., Tarricone, L., 2011. Green turtle (*Chelonia mydas*) in the north western Adriatic Sea: two interesting cases, in: Book of Abstracts. 4th Mediterranean Conference on Marine Turtles Naples - Italy. p. 58.

Van Buskirk, J., Crowder, L.B., 1994. Life-history variation in marine turtles. Copeia 1, 66–81.

Van de Elst, R., 2006. Status of leatherback turtles in Somalia, in: Hamann, M., Limpus, C., Hughes, G., Mortimer, J., Pilcher, N. (Eds.), Assessment of the Conservation Status of the Leatherback Turtle in the Indian Ocean and South-East Asia. IOSEA Species Assessment: Volume I. pp. 122–124.

Van de Giessen, E., 2011. Horn of Africa environmental security assessment.

Van de Merwe, J.P., Hodge, M., Olszowy, H. a, Whittier, J.M., Lee, S.Y., 2010. Using blood samples to estimate persistent organic pollutants and metals in green sea turtles (*Chelonia mydas*). Mar. Pollut. Bull. 60, 579–88.

Van de Merwe, J.P., Ibrahim, K., Whittier, J.M., 2005. Effects of hatchery shading and nest depth on the development and quality of *Chelonia mydas* hatchlings: implications for hatchery management in Peninsular; Malaysia. Aust. J. Zool. 53, 205–211.

Van Houtan, K.S., 2011. Assessing the impact of fishery actions to marine turtle populations in the North Pacific using classical and climate-based models. Internal Report IR-11-024, NOAA Fisheries, Pacific Islands Science Center, Honolulu, HI USA.

Van Houtan, K.S., Bass, A.L., 2007. Stormy oceans are associated with declines in sea turtle hatching. Curr. Biol. 17, R590–1.

Van Houtan, K.S., Halley, J.M., 2011. Long-Term Climate Forcing in Loggerhead Sea Turtle Nesting. PLoS One 6, e19043.

Van Houtan, K.S., Hargrove, S.K., Balazs, G.H., 2010. Land use, macroalgae, and a tumor-forming disease in marine turtles. PLoS One 5, e12900.

Van Houtan, K.S., Kittinger, J.N., 2014. Historical commercial exploitation and the current status of Hawaiian green turtles. Biol. Conserv. 170, 20–27.

Van Houtan, K.S., Kittinger, J.N., Lawrence, A.L., Yoshinaga, C., Born, V.R., Fox, A., 2012. Hawksbill sea turtles in the Northwestern Hawaiian Islands. Chelonian Conserv. Biol. 11, 117–121.

Van Piggelen, D.C.G., Strijbosch, H., 1993. The nesting of sea turtles (*Caretta caretta* and *Chelonia mydas*) in the Göksu Delta, Turkey, June-August, 1991. Turkish J. Zool. 17, 137–149.

Vander Zanden, H.B., Bjorndal, K.A., Bolten, A.B., 2013. Temporal consistency and individual specialization in resource use by green turtles in successive life stages. Oecologia 173, 767–77.

Vaughn, P.W., 1981. Marine turtles: a revies of their status and management in the Solomon Islands. Unpubl. Rep. to WWF 70 pp.

Venizelos, L., 2007. MEDASSET's Field Conservation Assessments in Turkey 1996-2007: A Retrospective, in: Proceedings of the Second National Symposium on Sea Turtles. Dalyan, Turkey, p. 5.

Venizelos, L., Kasparek, M., 2006. The white ghosts of Kazanli: sea turtle conservation in Turkey, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Book of Abstracts - 26th Annual Symposium on Sea Turtle Biology and Conservation. ISTS.

Venkatesan, S., Kannan, P., Rajagopalan, M., Vivekanandan, E., 2004. Nesting ecology of the green sea turtle *Chelonia mydas* along the Saurashtra coast. J. Mar. Biol. Assoc. India 46, 169–177.

Vera, V., 2008. Nesting of green turtles in Aves Island Wildlife Refuge. 2006 season, in: Rees, A.F., Frick, M.A., Panagopoulou, A., Williams, J.A. (Eds.), Proceedings of the Twenty-Seventh Annual Symposium on Sea Turtle Biology and Conservation. NOAA Tech Memo NMFS-SEFSC-569, Myrtle Beach, South Carolina, p. 275.

Vera, V., Buitrago, J., 2012. Actividad reproductiva de *Chelonia mydas* (Testudines: Cheloniidae) en Isla de Aves , Venezuela (2001-2008). Biol. Trop. 60, 745–758.

Vera, V., Guada, H.J., 2006. Aves Island Wildlife Refuge, Venezuela. Results 2002 peak nesting season, in: Pilcher, N.J. (Ed.), Proceedings of the 23rd Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-536, Kuala Lumpur, Malaysia, pp. 98–99.

Vera, V., Montilla, A., 2006. Results of sea turtle nesting on Aves Island, Venezuela: 2005 season, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Book of Abstracts. Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece, p. 331.

Viada, S.T., Hammer, R.A., Racca, R., Hannay, D., Thompson, M.J., Balcom, B.J., Phillips, N.W., 2008. Review of potential impacts to sea turtles from underwater explosive removal of offshore structures. Environ. Impact Assess. Rev. 2008 28, 267–285.

Videira, E.J.S., Pereira, M.A.M., Louro, C.M.M., 2011. Monitoring, tagging and conservation of marine turtles in Mozambique: annual report 2010/2011. Maputo, AICM/GTT.

Videira, E.J.S., Pereira, M.A.M., Louro, C.M.M., Narane, D.A., 2008. Monitoring, tagging and conservation of marine turtles in Mozambique: historical data and annual report 2007/08.

Viejobueno, S., Adams, C., Arauz, R., 2013. Conservacion e investigacion de last tortugas marinas en el Pacifico de Costa Rica. Reporte tecnico Julio 2012-Febrero 2013. Unpublished Report.

Villegas Barrutieta, J.A., 2000. Monitoring of marine turtle nesting in Isla Holbox, Quintana Roo, Mexico, 1997 season, in: Abreu-Grobois, F.A., Briseño-Dueñas, R., Márquez-Millán, R., Sarti-Martínez, L. (Eds.), Proceedings of the Eighteenth International Sea Turtle Symposium. NOAA Technical Memorandum MFS-SEFSC-436, Mazatlán, Sinaloa México, pp. 220–221.

Waayers, D., 2003. Nesting Distribution of Green Turtles (*Chelonia mydas*) and Loggerhead Turtles (*Caretta caretta*) between Gnaraloo Homestead and Carnarvon, including Bernier and Dorre Islands, Western Australia. Australia: Ecowaays Australia. Unpublished report.

Waayers, D., 2010. A Holistic Approach to Planning for Wildlife Tourism: A Case Study of Marine Turtle Tourism and Conservation in the Ningaloo Region, Western Australia. Murdoch University, Western Australia.

Waayers, D., Fitzpatrick, J., 2013. Genetic affiliations and key habitats of marine turtles in the Kimberly region, western Australia, in: Prince, R.I.T., Whiting, S., Raudino, H., Vitenbergs, A., Pendoley, K. (Eds.), Proceedings of the First Western Australian Marine Turtle Symposium 28-29th August 2012. Science Division, Department of Parks and Wildlife, Perth, Western Australia, pp. 34–36.

Wabnitz, C., Balazs, G.H., Beavers, S., Bjorndal, K.A., Bolten, A.B., Christensen, V., Hargrove, S., Pauly, D., 2010. Ecosystem structure and processes at Kaloko Honokohau, focusing on the role of herbivores, including the green sea turtle *Chelonia mydas*, in reef resilience. Mar. Ecol. Prog. Ser. 420, 27–44.

Wabnitz, C., Nichols, W.J., 2010. Editorial: Plastic Pollution: An Ocean Emergency. Mar. Turt. Newsl. 1–4.

Wainwright, T.C., Chilcote, M.W., Lawson, P.W., Nickelson, T.E., Huntington, C.W., Mills, J.S., Moore, K.M.S., Reeves, G.H., Stout, H.A., Weitkamp, L.A., 2008. Biological recovery criteria for the Oregon Coast coho salmon evolutionarily significant unit. U.S. Dept. Commer., NOAA Tech. Memo. NMFS-NWFSC-91.

Wallace, B.P., DiMatteo, A.D., Hurley, B.J., Finkbeiner, E.M., Bolten, A.B., Chaloupka, M.Y., Hutchinson, B.J., Abreu-Grobois, F.A., Amorocho, D.F., Bjorndal, K.A., Bourjea, J., Bowen, B.W., Dueñas, R.B., Casale, P., Choudhury, B.C., Costa, A., Dutton, P.H., Fallabrino, A., Girard, A., Girondot, M., Godfrey, M.H., Hamann, M., López-Mendilaharsu, M., Marcovaldi, M.Â., Mortimer, J.A., Musick, J.A., Nel, R., Pilcher, N.J., Seminoff, J.A., Troëng, S., Witherington, B.E., Mast, R.B., 2010a. Regional management units for marine turtles: a novel framework for prioritizing conservation and research across multiple scales. PLoS One 5, e15465.

Wallace, B.P., Kot, C.Y., DiMatteo, A.D., Lee, T., Crowder, L.B., Lewison, R.L., 2013. Impacts of fisheries bycatch on marine turtle populations worldwide: toward conservation and research priorities. Ecosphere 4.

Wallace, B.P., Lewison, R.L., McDonald, S.L., McDonald, R.K., Kot, C.Y., Kelez, S., Bjorkland, R.K., Finkbeiner, E.M., Helmbrecht, S., Crowder, L.B., 2010b. Global patterns of marine turtle bycatch. Conserv. Lett. 3, 131–142.

Wamukoya, G.M., Kaloki, F., Mbindo, C., 1996. The status of sea turtle conservation in Kenya, in: Humphrey, S.L., Salm, R.V. (Eds.), Status of Sea Turtle Conservation in the Western Indian Ocean. UNEP Regional Seas Reports and Studies No. 165. IUCN/UNEP, Nairobi, Kenya, pp. 57–72.

Wamukoya, G.M., Kaloki, F.P., Mbendo, J.R., 1997. Sea Turtle Recovery Action Plan for Kenya (STRAP). KESKOM Tech. Rep. Ser. 69.

Wang, Y., 2006. Sea turtle reserve and satellite tracking in China, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Proceedings of the Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece, p. 116.

Waples, R.S., McClure, M.M., Wainwright, T.C., McElhany, P., Lawson, P.W., 2010. Integrating evolutionary considerations into recovery planning for Pacific salmon, in: DeWoody, J.A., Bickham, J.W., Michler, C.H., Nichols, K.M., Rhodes, O.E., Woeste, K.E. (Eds.), Molecular Approaches in Natuarl Resource Conservation and Management. Cambridge University Press, New York, New York, pp. 239–266.

Waqas, U., Hasnain, S.A., Ahmad, E., Abbasi, M., Pandrani, A., 2011. Conservation of Green Turtle (*Chelonia mydas*) at Daran Beach, Jiwani, Balochistan. Pak. J. Zool. 43, 85–90.

Ward, M.W., Elkins, C., Ward, R.J., 2012. Punta Pargos green turtle monitoring and protection program of Costa Rica. 2011: Technical Report. MINAE ACT. SASP-PI-027-2009.

Waycott, M., Duarte, C.M., Carruthers, T.J.B., Orth, R.J., Dennison, W.C., Olyarnik, S., Calladine, A., Fourqurean, J.W., Heck, K.L., Hughes, A.R., Kendrick, G.A., Kenworthy, W.J., Short, F.T., Williams, S.L., 2009. Accelerating loss of seagrasses across the globe threatens coastal ecosystems. Proc. Natl. Acad. Sci. 106, 12377–12381.

Waycott, M., Longstaff, B.J., Mellors, J., 2005. Seagrass population dynamics and water quality in the Great Barrier Reef region: a review and future research directions. Mar. Pollut. Bull. 51, 343–350.

Weaver, M.L., 1996. Broadening environmental management in Fiji. Pacific Conserv. Biol. 2, 315–320.

Webb, A.P., Kench, P.S., 2010. The dynamic response of reef islands to sea-level rise: Evidence from multi-decadal analysis of island change in the Central Pacific. Glob. Planet. Change 72, 234–246.

Weber, N., Weber, S.B., Godley, B.J., Ellick, J., Witt, M., Broderick, A.C., 2013. Telemetry as a tool for improving estimates of marine turtle abundance. Biol. Conserv. 167, 90–96.

Weber, S.B., Broderick, A.C., Groothuis, T.G.G., Ellick, J., Godley, B.J., Blount, J.D., 2012. Fine-scale thermal adaptation in a green turtle nesting population. Proc. R. Soc. B 279, 1077–1084.

Wedding, L.M., Friedlander, A.M., 2008. Determining the influence of seascape structure on coral reef fishes in Hawaii using a geospatial approach. Mar. Geod. 31, 246–266.

Wedding, L.M., Friedlander, A.M., McGranaghan, M., Yost, R.S., Monaco, M.E., 2008. Using bathmetric lidar to define nearshore benthic habitat complexity: Implications for management of reef fish assemblages in Hawaii. Remote Sens. Environ. 112, 4159–4165.

Weijerman, M., van Tienen, L.H.G., Schouten, A.D., Hoekert, W.E.J., 1998. Sea turtles of Galibi, Suriname., in: Byles, R., Fernandez, Y. (Eds.), Proceedings of the Sixteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS- SEFSC-412, Hilton Head, South Carolina, pp. 142–144.

Weir, C.R., 2007. Observations of Marine Turtles in Relation to Seismic Airgun Sound off Angola. Assessment 95–96.

Weir, C.R., Ron, T., Morais, M., Duarte, A.D.C., 2007. Nesting and at-sea distribution of marine turtles in Angola, West Africa, 2000–2006: occurrence, threats and conservation implications. Oryx 41, 224.

Weishampel, J.F., Bagley, D.A., Ehrhart, L.M., 2004. Earlier nesting by loggerhead sea turtles following sea surface warming. Glob. Chang. Biol. 10, 1424–1427.

Weishampel, J.F., Bagley, D.A., Ehrhart, L.M., Weishampel, A., 2010. Nesting phenologies of two sympatric sea turtle species related to sea surface temperatures. Endanger. Species Res. 12, 41–47.

Wenninger, P., 2012. Annual report, sea turtle nesting: military lease area, Tinian CNMI, Fiscal Year 2012 (Oct. 1, 2011-Sept. 30, 2012).

Wershoven, J.L., Wershoven, R.W., 1992. Juvenile green turtles in their nearshore habitat of Broward County, Florida: a five year review, in: Salmon, M., Wyneken, J. (Eds.), Proceedings of the Eleventh Annual Workshop on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFC-302, pp. 121–123.

Western and Central Pacific Fisheries Commission, 2008. 2007 national tuna fishery report - Solomon Islands.

Wetherall, J.A., 2012. Estimated number of green turtles nesting at East Island, French Frigate Shoals, 1973-2012. Data Document.

Wetherall, J.A., Balazs, G.H., Tokunaga, R.A., Yong, M.Y.Y., 1993. Bycatch of marine turtles in the North Pacific high-seas drift-net fisheries and impacts on the stocks. Int. North Pacific Fish. Comm. Bull. 53, 519–538.

Wetterer, J.K., 2006. The tropical fire ant, Solenopsis geminata, on an impotent sea turtle nesting beach in Tortuguero National Park, Costa Rica, in: Pilcher, N.J. (Ed.), Proceedings of the Twenty-Third Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-536, pp. 259–261.

White, M., 2012a. Sea Turtles in the Cook Islands. Volume One: 2009-2012.

White, M., 2012b. An Assessment of sea turtles at Palmerston Atoll, Southern Cook Islands. Rufford Trust, Rarotonga, Cook Islands.

White, M., 2013. The first study of sea turtles at Rarotonga, Southern Cook Islands. Testudo 7, 12–29.

White, M., Galbraith, G., 2013. Rakahanga Atoll: sea turtles at a remote site in Oceania. Testudo 7, 30–48.

Whiting, S.D., Guinea, M., Pike, G.D., 2000. Sea turtle nesting in the Australian Territory of Ashmore and Cartier Islands, Eastern Indian Ocean. Sea turtles Indo-Pacific Res. Manag. Conserv. Asean Acad. Press. London 86–93.

Whittow, G.C., Balazs, G.H., 1982. Basking behavior of the Hawaiian green turtle (*Chelonia mydas*). Pacific Sci. 36, 129–139.

Williams, E.H., Bunkley-Williams, L., 2000. Marine major ecological disturbances of the Caribbean. Infect. Dis. Revies 2, 110–127.

Williams, S.L., 1988. Assessment of Anchor Damage and Carrying Capacity of Seagrass Beds in Francis and Mayo Bays for Green Sea Turtles. Biosph. Reserv. Rep. No. 25, U.S. Dept. Inter. Natl. Park Serv. Virgin Islands Resour. Manag. Coop. Virgin Islands Natl. Park 32 pp.

Wilson, L., MacKay, K.T., Trevor, A.P., Solomona, P., 2004. Melanesian Marine Turtles Conservation Forum workshop report. Western Pacific Regional Fishery Management Council (WPRFMC); the Secretariat of the Pacific Regional Environment Programme (SPREP); the Canada- South Pacific Ocean Development (C-SPOD) Program and WWF South Pacific, Gizo, Solomon Islands.

Win, A.L., Lwin, M.M., 2012. Interaction between fisheries activities and marine turtles in Myanmar, in: Proceedings of the 7th International Symposium on SEASTAR2000 and Asian Bio-Logging Science (The 11th SEASTAR2000 Workshop). pp. 5–7.

Witherington, B.E., 1986. Human and natural causes of marine turtle clutch and hatcling mortality and their relationship to hatcling production on an important Florida nesting beach. Unpublished Master of Science thesis. University of Central Florida, Orlando, Florida.

Witherington, B.E., 1992. Behavioral responses of nesting sea turtles to artificial lighting. Herpetologica 48, 31–39.

Witherington, B.E., 1997. The problem of photopollution for sea turtles and other nocturnal animals, in: Clemmons, J.R., Buchholz, R. (Eds.), Behavioral Approaches to Conservation in the Wild. Cambridge University Press, pp. 303–328.

Witherington, B.E., 1999. Reducing threats to nesting habitat, in: Eckert, K.L., Bjorndal, K.A., Abreu-Grobois, F.A., Donnelly, M. (Eds.), Research and Management Techniques for the Conservation of Sea Turtles. IUCN/SSC Marine Turtle Specialist Group Publication No. 4, pp. 179–183.

Witherington, B.E., 2002. Ecology of neonate loggerhead turtles inhabiting lines of downwelling near a Gulf Stream front. Mar. Biol. 140, 843–853.

Witherington, B.E., Bjorndal, K.A., 1991. Influences of artificial lighting on the seaward orientation of hatchling loggerhead turtles *Caretta caretta*. Biol. Conserv. 55, 139–149.

Witherington, B.E., Bresette, M.J., Herren, R., 2006. *Chelonia mydas* – green Turtle, in: Meylan, P.A. (Ed.), Biology and Conservation of Florida Turtles. Chelonian Research Monographs 3:90-104.

Witherington, B.E., Ehrhart, L.M., 1989a. Hypothermic stunning and mortality of marine turtles in the Indian River Lagoon system, Florida. Copeia 1989, 696–703.

Witherington, B.E., Ehrhart, L.M., 1989b. Status and reproductive characteristics of green turtles (*Chelonia mydas*) nesting in Florida, in: Ogren, L., Berry, F., Bjorndal, K., Kumpf, H., Mast, R., Medina, G., Reichart, H., Witham, R. (Eds.), Proceedings of the Second Western Atlantic Turtle Symposium. NOAA Technical Memorandum NMFS-SEFC-226, pp. 351–352.

Witherington, B.E., Hirama, S., Hardy, R., 2012. Young sea turtles of the pelagic Sargassum-dominated drift community: habitat use, population density, and threats. Mar. Ecol. Prog. Ser. 463, 1–22.

Witherington, B.E., Koeppel, C.M., 2000. Sea turtle nesting in Florida, USA, during the decade 1989-1998: an analysis of trends, in: Kalb, H.J., Wibbels, T. (Eds.), Proceedings of the Nineteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-443, pp. 94–96.

Witherington, B.E., Lucas, L., Koeppel, C., 2005. Nesting sea turtles respond to the effects of ocean inlets, in: Coyne, M.S., Clark, R.D. (Eds.), Proceedings of the Twenty-First Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-528, pp. 355–356.

Witherington, B.E., Martin, R.E., 1996. Understanding, assessing, and resolving light-pollution problems on sea turtle nesting beaches, Environmental Protection.

Witmer, G.W., Boyd, F., Hillis-Starr, Z.M., 2007. The successful eradication of introduced roof rats (Rattus rattus) from Buck Island using diphacinone, followed by an irruption of house mice (Mus musculus). Wildl. Res. 34, 108–115.

Witsen, D., 2012. Djibouti biodiversity and tropical forests : 118/119 assessment.

Witzell, W.N., 1982. Observations on the Green Sea Turtle (*Chelonia mydas*) in Western Samoa. Copeia 1982, 183–185.

Witzell, W.N., 1994. The U.S. commercial sea turtle landings. NOAA Technical Memorandum NMFS-SEFSC-350.

Wood, F.E., Wood, J.R., 1994. Sea turtles of the Cayman Islands, in: Brunt, M.A., Davies, J.A. (Eds.), The Cayman Islands: Natural History and Biogeography. Kluwer Academic Press, Netherlands, pp. 229–236.

Woodrom Rudrud, R., 2010. Forbidden sea turtles: Traditional laws pertaining to sea turtle consumption in Polynesia (Including the Polynesian Outliers). Conserv. Soc. 8, 84–97.

Work, T.M., Balazs, G.H., Schumacher, J.L., Marie, A., 2005. Epizootiology of spirorchiid infection in green turtles (*Chelonia mydas*) in Hawaii. J. Parasitol. 91, 871–876.

Wright, B., Mohanty, B., 2002. Olive ridley mortality in gill nets in Orissa. Kachhapa 6, 18.

Wright, L.I., Fuller, W.J., Godley, B.J., McGowan, A., Tregenza, T., Broderick, A.C., 2012a. Reconstruction of paternal genotypes over multiple breeding seasons reveals male green turtles do not breed annually. Mol. Ecol.

Wright, L.I., Stokes, K.L., Fuller, W.J., Godley, B.J., McGowan, A., Snape, R., Tregenza, T., Broderick, A.C., 2012b. Turtle mating patterns buffer against disruptive effects of climate change. Proc. Biol. Sci. 279, 2122–2127.

Wusstig, S., 2012. Haggan: green sea turtle [WWW Document]. URL http://guampedia.com/haggan-green-sea-turtle (accessed 4.23.13).

Wyneken, J., Balazs, G.H., 1996. Morphological differences in hind flippers of hatchling green turtles from the Hawaiian Islands and Florida, U.S.A., in: Keinath, J.A., Barnard, D.E., Musick, J.A., Bell, B.A. (Eds.), Proceedings of the Fifteenth Annual Symposium on Sea

Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-387, pp. 348–349.

Wyneken, J., Salmon, M., 1992. Frenzy and postfrenzy swimming activity in loggerhead, green and leatherback hatchling sea turtles. Copeia 2, 478–484.

Xavier, R., Barata, A., Cortez, L.P., Queiroz, N., Cuevas, E., 2006. Hawksbill turtle (*Eretmochelys imbricata* Linnaeus 1766) and green turtle (*Chelonia mydas* Linnaeus 1754) nesting activity (2002-2004) at El Cuyo beach,Mexico beach, Mexico. Amphibia-Reptilia 27, 539–547.

Yalçın-Özdilek, Ş., 2007. Status of sea turtles (*Chelonia mydas* and *Caretta caretta*) on Samandaâ Beach, Turkey: a five-year monitoring study. Ann. Zool. Fennici 44, 333–347.

Yalçın-Özdilek, Ş., Aureggi, M., 2006. Strandings of Juvenile Green Turtles at Samandağ , Turkey. Chelonia Conserv. Biol. 5, 152–154.

Yalçın-Özdilek, Ş., Kaska, Y., Sönmez, B., 2009. Preliminary study on sand and green turtle nest temperatures and sex ratio of hatchlings on Samandag Beach, Turkey, in: Demetropoulos, A., Turkozan, O. (Eds.), Proceedings of the Second Mediterranean Conference on Marine Turtles. Barcelona Convention – Bern Convention – Bonn Convention (CMS), Kemer, Turkey, p. 179.

Yalçın-Özdilek, Ş., Sönmez, B., 2011. Nesting Characteristics at Samandağ and Extended Beaches, Turkey. Mar. Turt. Newsl. 131, 7–9.

Yalçın-Özdilek, Ş., Yalçın, S., 2012. Wind Energy Plants and Possible Effects on Samandağ Sea Turtles. Mar. Turt. Newsl. 133, 7–9.

Yasuda, T., Tanaka, H., Kittiwattanawong, K., Mitamura, H., Klom-in, W., Arai, N., 2006. Do female green turtles (*Chelonia mydas*) exhibit reproductive seasonality in a year-round nesting rookery? J. Zool. 269, 451–457.

Yerli, S. V., Canbolat, A.F., 1998. Results of a 1996 survey of Chelonia in Turkey. Mar. Turt. Newsl. 9–11.

Yilmaz, C., Türkozan, O., Oruç, A., 2009. Tuzla, Akyatan ve Yumurtalık Milli Parkı Kumsalları Deniz Kaplumbağası (*Chelonia mydas* ve *Caretta caretta*) ve Yumuşak Kabuklu Nil Kaplumbağası (Trionyx triunguis) Popülasyonlarının Araştırılması, İzlenmesi ve Korunması Çalışması, Kasım 2009, Edebiyat A Journal Of Middle Eastern Literatures. Istanbul, Turkey.

Zainudin, I.M., Pet-Doede, L., Hitipeuw, C., Adnyana, I.B.W., 2008. Interaction of sea turtles with Indonesia fisheries – preliminary findings [WWW Document]. Indian Ocean – South-east Asian Mar. Turt. Memo. Underst. Profile Mon. URL http://www.ioseaturtles.org/pom_detail.php?id=76

Zambrano, R., Rodriguez, E., 1995. The nesting biology of hawksbill (*Eretmochelys imbricata*) and green (*Chelonia mydas*) sea turtles in El Cuyo, Yucatan; Mexico, in: Richardson, J.I., Richardson, T.H. (Eds.), Proceedings of the Twelfth Annual Workshop on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-361, pp. 252–256.

Zárate, P., 2012. Offshore Oasis: Ecology of Sea Turtles at Oceanic Islands of the Eastern Pacific, in: Seminoff, J., Wallace, B. (Eds.), Sea Turtles of the Eastern Pacific. University of Arizona Press, Tucson, Arizona, pp. 65–87.

Zárate, P., Bjorndal, K.A., Parra, M., Dutton, P.H., Seminoff, J.A., Bolten, A.B., 2013. Hatching and emergence success in green turtle *Chelonia mydas* nests in the Galápagos Islands. Aquat. Biol. 19, 217–229.

Zárate, P., Cahoon, S.S., Contato, M.C.D., Dutton, P.H., Seminoff, J.A., 2006. Nesting beach monitoring of green turtles in the Galapagos Islands: a 4-year evaluation, in: Frick, M., Panagopoulou, A., Rees, A.F., Williams, K. (Eds.), Book of Abstracts. Twenty Sixth Annual Symposium on Sea Turtle Biology and Conservation. International Sea Turtle Society, Athens, Greece, p. 337.

Zárate, P., Fernie, A., Dutton, P.H., 2003. First results of the East Pacific green turtle, *Chelonia mydas*, nesting population assessment in the Galapagos Islands, in: Seminoff, J.A. (Ed.), Twenty- Second Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503, Miami, FL, pp. 70–73.

Zug, G.R., Balazs, G.H., Wetherall, J.A., Parker, D.M., Murakawa, S.K.K., 2002. Age and growth of Hawaiian green seaturtles (*Chelonia mydas*): an analysis based on skeletochronology. Fish. Bull. 100, 117–127.

Zug, G.R., Glor, R.E., 1998. Estimates of age and growth in a population of green sea turtles (*Chelonia mydas*) from the Indian River lagoon system, Florida: a skeletochronological analysis. Can. J. Zool. 76, 1497–1506.

Zurita, J.C., Herrera P., R., Arenas, A., Negrete, A.C., Gómez, L., Prezas, B., Sasso, C.R., 2012. Age at first nesting of green turtles in the Mexican Caribbean, in: Jones, T.T., Wallace, B.P. (Eds.), Proceedings of the 31st Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NOAA NMFS-SEFSC-631, p. 75.

Zurita, J.C., Herrera P., R., Arenas, A., Torres, M.E., Calderón, C., Gómez, L., Alvarado, J.C., Villavicencio, R., 2003. Nesting loggerhead and green sea turtles in Quintana Roo, Mexico, in: Seminoff, J.A. (Ed.), Proceedings of the 22nd Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-503, Miami, Florida, pp. 125–127.

Zurita, J.C., Herrera, R., Prezas, B., 1993. Tortufas marinas del Caribe, in: Salazar-Vallejo, S.I., Gonzalez, N.E. (Eds.), Biodiversidad Marina Y Costera de Mexico. Com. Nal. Biodiversidad y CIQRO, Mexico City, pp. 735–750.

Zurita, J.C., Prezas, B., Herrera P., R., Miranda, J.L., 1994. Sea turtle tagging program in Quintana Roo, Mexico, in: Bjorndal, K.A., Bolten, A.B., Johnson, D.A., Eliazar, P.J. (Eds.), Proceedings of the Fourteenth Annual Symposium on Sea Turtle Biology and Conservation. NOAA Technical Memorandum NMFS-SEFSC-351, Hilton Head, South Carolina, pp. 300–303.

Appendix 1. References for demographic parameters presented in Table 2.1.

Nesting Site	Mean Nesting Size (SCL, cm)	Remigration Interval (yrs.)	Nesting Frequency (nests/yr)	Clutch Size (eggs/clutch)	Survival Rates	Growth Rates (SCL; cm/year)	Age at First Reproduction (years)	Sex Ratio
DPS 1: NORTH ATLANTIC								
Archie Carr National Wildlife Refuge, Florida, USA	Witherington and Ehrhart, 1989b	Witherington and Ehrhart, 1989b	Johnson and Ehrhart, 1996	Witherington and Ehrhart, 1989b	Witherington *et al.*, 2006	Witherington and Ehrhart, 1989b	Frazer and Ehrhart, 1985	Schroeder and Owens, 1994
Core Index Beaches, FL, USA	–	–	Florida Fish and Wildlife Conservation Commission, 2012b	Florida Fish and Wildlife Conservation Commission, 2012b	–	–	–	–
El Cuyo, Yucatan, Mexico	–	–	Xavier *et al.*, 2006	Xavier *et al.*, 2006	–	–	–	–
Isla Holbox, Quintana Roo, Mexico	–	Zurita *et al.*, 1994	–	Villegas Barrutieta, 2000	–	–	Zurita *et al.*, 2012	–
Central Coast, Quintana Roo, Mexico	–	Zurita *et al.*, 1994	–	Agardy and Gil Hernandez, 1989	Arenas *et al.*, 2007	–	Zurita *et al.*, 2012	–
Isla Aguada, Campeche, Mexico	–	–	Guzmán-Hernández, 2001, 2002, 2003, 2005, 2006a, 2006b, 2006c; Guzmán-Hernández *et al.*, 2008; Guzmán-Hernández and García Alvarado, 2009, 2010, 2011	Guzmán-Hernández, 2001, 2002, 2003, 2005, 2006b; Guzmán-Hernández *et al.*, 2008; Guzmán-Hernández and García Alvarado, 2009, 2010, 2011	Guzmán-Hernández, 2005; Guzmán-Hernández *et al.*, 2008	–	–	–
Tortuguero, Costa Rica	Carr and Ogren, 1960	Troëng and Chaloupka, 2007	Carr *et al.*, 1978	Tiwari *et al.*, 2006	Troëng and Chaloupka, 2007	–	Frazer and Ladner, 1986	Spotila *et al.*, 1987; Witherington *et al.*, 2006
DPS 2: MEDITERRANEAN								
Akyantan, Turkey	–	–	–	O. Türkozan, Adnan Menderes Universitesi, Turkeypers. comm., 2011	–	–	–	Casale *et al.*, 2000
Kazanli, Turkey	Baran *et al.*, 1991	–	–	Aureggi, 2001	–	–	–	–
Samadang, Turkey	–	–	–	Yalçın-Özdilek, 2007	–	–	–	Yalçın-Özdilek *et al.*, 2009
Alagadi, Cyprus	Broderick and Godley, 1994	Broderick *et al.*, 2003	Broderick *et al.*, 2003	Broderick *et al.*, 2003	–	Broderick *et al.*, 2003	–	Broderick *et al.*, 2000; Wright *et al.*, 2012a

Nesting Site	Mean Nesting Size (SCL, cm)	Remigration Interval (yrs.)	Nesting Frequency (nests/yr)	Clutch Size (eggs/clutch)	Survival Rates	Growth Rates (SCL; cm/year)	Age at First Reproduction (years)	Sex Ratio
West Coast, Cyprus	–	Demetropoulos and Hadjichristophorou, 1989	Demetropoulos and Hadjichristophorou, 1989	Demetropoulos and Hadjichristophorou, 1989	–	–	–	Broderick *et al.*, 2000
Israel	–	Y. Levy, Israel Sea Turtle Rescue Centre, pers. comm., 2012	–	Y. Levy, Israel Sea Turtle Rescue Centre, pers. comm., 2012	–	–	–	–
Lattakia Beach, Syrian Arab Republic	Rees *et al.*, 2008	–	–	Rees *et al.*, 2008	–	–	–	–
Ras al-Bassit, Syrian Arab Republic	–	–	–	Hirth, 1997	–	–	–	–
Wadi Kandil Beach, Syrian Arab Republic	–	–	–	Hirth, 1997	–	–	–	–
DPS 3: SOUTH ATLANTIC								
Bioko Island, Equatorial Guinea	–	–	Tomás *et al.*, 2010	Tomás *et al.*, 2000	–	–	–	–
Poilão Bijagos Archipelago, Guinea Busseau	–	–	–	Limoges and Robillard, 1991; Catry *et al.*, 2002, 2009, 2010	–	–	–	Catry *et al.*, 2010; Rebelo *et al.*, 2011
Ascension Island, UK	–	Mortimer and Carr, 1987	Mortimer and Carr, 1987	Mortimer and Carr, 1987	–	–	Frazer and Ladner, 1986	Broderick *et al.*, 2001; Godley *et al.*, 2002a; Pintus *et al.*, 2009
Aves Island, Venezuela	Rainey, 1971; Vera and Guada, 2006; Vera, 2008; Prieto *et al.*, 2012; Vera and Buitrago, 2012	Sole and Medina, 1989; Prieto *et al.*, 2012	Sole and Medina, 1989; Prieto *et al.*, 2012; Vera and Buitrago, 2012	Cruz *et al.*, 2010; Prieto *et al.*, 2012	–	–	–	–
Galibi Reserve, Suriname	Schulz, 1975	Schulz, 1975; Weijerman *et al.*, 1998	Schulz, 1975; Weijerman *et al.*, 1998	Schulz, 1975	–	–	Frazer and Ladner, 1986	Mrosovsky *et al.*, 1984; Godfrey *et al.*, 1996
Trindade Island, Brazil	Moreira *et al.*, 1995; Almeida *et al.*, 2011	Almeida *et al.*, 2011	Almeida *et al.*, 2011	Almeida *et al.*, 2011	–	–	–	–
Atol das Rocas, Brazil	Bellini *et al.*, 2013	Bellini *et al.*, 2013	Bellini *et al.*, 2013	Bellini *et al.*, 2013	–	–	–	–
DPS 4: SOUTHWEST INDIAN								
Aldabra, Seychelles Islands	Fraizer, 1971	–	Hirth, 1997	Hirth, 1997	–	–	–	–

Nesting Site	Mean Nesting Size (SCL, cm)	Remigration Interval (yrs.)	Nesting Frequency (nests/yr)	Clutch Size (eggs/clutch)	Survival Rates	Growth Rates (SCL; cm/year)	Age at First Reproduction (years)	Sex Ratio
Mohéli, Comoros Islands	Frazier, 1985	–	–	Innocenzi *et al.*, 2010	–	–	–	Innocenzi *et al.*, 2010
Mayotte, Comoros Islands, France	Frazier, 1985	Bourjea *et al.*, 2007a	Bourjea *et al.*, 2007a	Frazier, 1985	–	–	–	–
Tromelin, Esparces Islands, France	Hughes, 1974	Hirth, 1997	Hirth, 1997	Hirth, 1997	–	–	–	–
Europa, Esparces Islands, France	Servan 1976 as cited in Groombridge and Luxmoore, 1989	Hirth, 1997	Hirth, 1997	Hirth, 1997	–	–	–	–
DPS 5: NORTH INDIAN								
Gujarat, India	–	–	–	Venkatesan *et al.*, 2004	–	–	–	–
Hawkes Bay and Sandpit, Pakistan	–	–	–	Minton, 1966	–	–	–	–
Sharma, Peoples Democratic Republic of Yemen	Hirth and Carr, 1970	–	–	Hirth, 1997	–	–	–	–
Ras al Hadd, Oman	Ross and Barwani, 1982	–	–	Mendonça *et al.*, 2010	–	–	–	Hasbún *et al.*, 2000
Ras Baridi, Saudi Arabia	Miller, 1989	–	–	Al-Merghani *et al.*, 2000	–	–	–	–
Karan and Jana Islands, Arabian Gulf, Saudi Arabia	Miller, 1989	–	Hirth, 1997	Hirth, 1997	–	–	–	–
DPS 6: EAST INDIAN/WEST PACIFIC								
Sarawak, Malaysia	–	Hendrickson, 1958	Hendrickson, 1958	Hendrickson, 1958	–	–	–	Leh, 1994; Tiwol and Cabanban, 2000
Redang Island, Malaysia	–	Caverhill *et al.*, 2012	Caverhill *et al.*, 2012	Caverhill *et al.*, 2012	–	–	–	Mortimer, 1991b; van de Merwe *et al.*, 2005
Sipadan, Sabah, Malaysia	–	–	–	–	–	–	–	–
Sabah Turtle Islands, Malaysia	–	Pilcher and Basintal, 2000	Pilcher and Basintal, 2000	Pilcher and Basintal, 2000	–	Pilcher and Basintal, 2000; Pilcher, 2010	–	Tiwol and Cabanban, 2000; Pilcher, 2010
Berau Islands, Berawan Archipelago, Indonesia	–	Reischig *et al.*, 2012	Adnyana *et al.*, 2008	Adnyana *et al.*, 2008	–	–	Reischig *et al.*, 2012	–
Enu Island (Aru Islands), Indonesia	–	–	Dethmers, 2010	Dethmers, 2010	–	–	–	–

Nesting Site	Mean Nesting Size (SCL, cm)	Remigration Interval (yrs.)	Nesting Frequency (nests/yr)	Clutch Size (eggs/clutch)	Survival Rates	Growth Rates (SCL; cm/year)	Age at First Reproduction (years)	Sex Ratio
Viet Nam	–	*Sea turtles education booklet, Viet Nam*, 2012	*Sea turtles education booklet, Viet Nam*, 2012	*Sea turtles education booklet, Viet Nam*, 2012	Ai, 2001	–	–	–
Turtle Islands, Philippines	Trono, 1991	Trono, 1991	Burton, 2012	Trono, 1991; Burton, 2012	Burton, 2012	–	–	Trono, 1991
Lanyu, Taiwan, Provence of China	–	Cheng *et al.*, 2008	Cheng *et al.*, 2008	Cheng *et al.*, 2008	–	–	–	King *et al.*, 2013
Wan-an, Taiwan, Provence of China	–	Cheng *et al.*, 2008	Cheng *et al.*, 2008	Cheng *et al.*, 2008; Chen *et al.*, 2010	–	–	–	King *et al.*, 2013
Thameehla Island, Myanmar	–	Lwin, 2009b	–	Lwin, 2009a	–	–	–	–
Pangumbahan, Java, Indonesia	–	–	–	Suwelo, 1971	–	–	–	–
Sukamade, Java, Indonesia	–	–	Hirth, 1997	Hirth, 1997	–	–	–	–
Western Australia	–	Prince, 1993	Miller *et al.*, 1997	–	–	–	–	–
Sri Lanka	–	Ekanayake *et al.*, 2004	Ekanayake *et al.*, 2004	Ekanayake and Ranawana, 2001a	–	–	–	Ekanayake and Ranawana, 2001b
DPS 7: CENTRAL WEST PACIFIC								
Ogasawara Islands, Japan	–	Abe *et al.*, 2003	Suganuma *et al.*, 1996	Suganuma *et al.*, 1996	–	–	–	–
DPS 8: SOUTHWEST PACIFIC								
Heron Island, southern Great Barrier Reef, Australia	Bustard, 1974	Limpus, 2009	Limpus, 2009	Limpus, 2009	Chaloupka and Limpus, 2005	Limpus, 2009	Chaloupka *et al.*, 2004	Limpus, 2009
Raine Island, northern Great Barrier Reef, Australia	Stoddart *et al.*, 1981	Limpus, 2009	Limpus *et al.*, 2003	Limpus *et al.*, 2003	Limpus *et al.*, 2003	Limpus *et al.*, 2003	Chaloupka *et al.*, 2004	Chaloupka *et al.*, 2004
DPS 9: CENTRAL SOUTH PACIFIC								
Rose Atoll, American Samoa	K. Van Houtan, NMFS, unpub.. data 2013	–	–	–	–	–	–	–
DPS 10: CENTRAL NORTH PACIFIC								
French Frigate Shoals, USA	Balazs, 1980	Balazs and Chaloupka, 2004a	Tiwari *et al.*, 2010	Balazs, 1980	–	Balazs and Chaloupka, 2004b	Balazs and Chaloupka, 2004a	Balazs, 1980

Nesting Site	Mean Nesting Size (SCL, cm)	Remigration Interval (yrs.)	Nesting Frequency (nests/yr)	Clutch Size (eggs/clutch)	Survival Rates	Growth Rates (SCL; cm/year)	Age at First Reproduction (years)	Sex Ratio
DPS 11: EAST PACIFIC								
Baja California, Mexico	Alvarado-Díaz and Figueroa, 1990	Márquez-Millán *et al.*, 1982; Alvarado-Díaz and Figueroa, 1990	Alvarado-Díaz *et al.*, 2003	Alvarado-Díaz and Figueroa, 1990	Seminoff *et al.*, 2003	Koch *et al.*, 2007; Delgado-Trejo, 2012	Alvarado-Díaz and Figueroa, 1990	Alvarado-Díaz and Figueroa, 1990
Galapagos Islands, Ecuador	Pritchard, 1971	Zárate *et al.*, 2006	Zárate *et al.*, 2003	Zárate *et al.*, 2003	–	Green, 1993	–	–

Appendix 2. Summary of data sources for green turtle nesting abundance data presented in the report.

Country	Nesting Site	Years	Reference
DPS 1: NORTH ATLANTIC			
Cayman Islands	Grand Cayman	2005-2009	Bell *et al.*, 2007; Dow *et al.*, 2007; Solomon and Blumenthal, 2008, 2009; Echternacht *et al.*, 2011; Cayman Islands Department of Environment, unpublished data
Cayman Islands	Little Cayman	2007	Bell *et al.*, 2007; Dow *et al.*, 2007; Solomon and Blumenthal, 2009; Echternacht *et al.*, 2011
Costa Rica	Tortuguero	1971-2011	Chaloupka *et al.*, 2008; Sea Turtle Conservancy, 2013
Cuba	Beaches of the Guahanacabibes Peninsula	1998-2010	Azanza *et al.*, 2013; Azanza Ricardo *et al.*, 2013a, 2013b
Cuba	Cayo Largo (Eastern Keys of Isla de la Juventud)	2001-2010	Dow *et al.*, 2007; Blanco *et al.*, 2009; Moncada *et al.*, 2010, 2011; Azanza Ricardo *et al.*, 2013a
Cuba	Cayo Rosario	2008	Blanco *et al.*, 2009
Cuba	Cayo Siju, Cayo Real, Juan Garcia (Cayos de San Felipe)	2007-2009	Dow *et al.*, 2007; Blanco *et al.*, 2009
Cuba	Eastern Keys of Isla de la Juventud	2010	Moncada *et al.*, 2010
Cuba	Guanal	1988-2011	Dow *et al.*, 2007; Blanco *et al.*, 2009; Moncada *et al.*, 2010, 2011
Cuba	Playas Archipiélago Jardines de la Reina	2011	Dow *et al.*, 2007; Blanco *et al.*, 2009; Moncada *et al.*, 2010, 2011
Cuba	San Felipe	2001-2011	Dow *et al.*, 2007; Blanco *et al.*, 2009; Moncada *et al.*, 2010, 2011
Cuba	South Isla de la Juventud	2010-2011	Dow *et al.*, 2007; Blanco *et al.*, 2009; Moncada *et al.*, 2010, 2011
Mexico	Campeche	1992-2012	Guzmán-Hernández *et al.*, 1993, 1994, 1995a, 1995b, 2008; Guzmán-Hernández, 1996, 1997, 2000, 2001, 2002, 2003, 2005, 2006c; Guzmán-Hernández and García Alvarado, 2011, 2013a, 2013b, 2009, 2010
Mexico	Quintana Roo	2010-2012	Julio Zurita, Univ. Quintana Roo, pers. comm. 2012
Mexico	Tamaulipas	1995-2000	R. Marquez-Millan pers. comm in Seminoff *et al.*, 2004
Mexico	Veracruz	1995-2000	R. Marquez-Millan pers. comm in Seminoff *et al.*, 2004
Mexico	Yucatán	1995-2000	R. Marquez-Millan pers. comm in Seminoff *et al.*, 2004
Nicaragua	El Cocal	2000	Lagueux and Campbell, 2005
Puerto Rico	Humacao	2012	Carlos Diez, pers. comm. 2013
Puerto Rico	Mona Island	2012	Carlos Diez, pers. comm. 2013
Puerto Rico	Vieques	2010-2012	Department of Environmental and Natural Resources of Puerto Rico, 2012

USA, FL	Brevard	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Broward	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Charlotte	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Collier	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Dry Tortugas National Park	2010-2011	Hart *et al.*, 2013
USA, FL	Duval	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Escambia	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Flagler	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Franklin	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Indian River	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Lee	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Manatee	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Martin	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Miami-Dade	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Monroe	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Nassau	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Okaloosa	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Palm Beach	2009-2010	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Sarasota	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	St. Johns	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	St. Lucie	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Volusia	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, FL	Walton	2008-2012	Florida Fish and Wildlife Conservation Commission, 2012a
USA, GA	Georgia	2010-2012	M. Dodd, GA DNR, pers. comm., 2013
USA, NC	North Carolina	2010	M. Godfrey, NC WRC, pers. comm., 2013
USA, SC	South Carolina	2010-2012	M. Pate, SC DNR, pers. comm., 2013
USA, TX	Texas	2010-2012	D. Shaver, National Park Service, pers. comm., 2012
DPS 2: MEDITERRANEAN			
Cyprus Region A	North Karpaz, Region A	1998-2002	Senol, 1999; Kasparek *et al.*, 2001; Senol and Kusetogulları, 2002
Cyprus Region A	Alagadi, Region A	1992-2000	Broderick and Godley, 1994;

			Broderick *et al.*, 2002a; Glen *et al.*, 2005
Cyprus Region A	South Karpaz, Region A	2001-2002	Kasparek *et al.*, 2001; Senol, 2001; Senol and Kusetogulları, 2002
Cyprus Region A	West Coast, Region A	1993-2007	Fuller *et al.*, 2010b
Cyprus Region B	West Coast, Region B (5 beaches)	1998-2008	Demetropoulos and Hadjichristophorou, 2010
Egypt	Egypt	1998	Clarke *et al.* 2000 cited in Nada and Casale, 2010
Greece	Greece	2007	Margaritoulis and Panagopoulou, 2010
Israel	Israel	1993-2008	Y. Levy, Israel Sea Turtle Rescue Centre, pers. comm., 2012
Lebanon	El Aabbassiye	2003-2005	http://issuu.com/medasset/docs/lebanon_report_2005
Lebanon	Tyre Coast Nature Reserve	2004-2005	http://issuu.com/medasset/docs/lebanon_report_2005
Lebanon	El Mansouri	2002-2005	http://issuu.com/medasset/docs/lebanon_report_2005
Syria	Latakia	2004-2009	Rees *et al.*, 2010
Syria	Ras el Basit	5 yrs; 2004-2009	Rees *et al.*, 2010
Syria	Um Toyour	3 yrs; 2004-2009	Rees *et al.*, 2010
Syria	Wadi Kandil	2004-2009	Rees *et al.*, 2010
Syria	Banias area	2004-2009	Rees *et al.*, 2010
Turkey	Alata*	2002-2006	Türkozan and Kaska, 2010
Turkey	Kazanli*	1988-2006	Türkozan and Kaska, 2010
Turkey	Akyatan	1988-2006	Türkozan and Kaska, 2010
Turkey	Sugozu	2004	Türkozan and Kaska, 2010
Turkey	Samandag*	1988-2010	Türkozan and Kaska, 2010; Yalçın-Özdilek and Sönmez, 2011
Turkey	Patara	2001	Türkozan and Kaska, 2010
Turkey	Fenike-Kumluca	1994	Türkozan and Kaska, 2010
Turkey	Belek	1994-2006	Türkozan and Kaska, 2010
Turkey	Kizilot	1990	Türkozan and Kaska, 2010
Turkey	Anamur	2006	Türkozan and Kaska, 2010
Turkey	Goksu Delta	1991-2006	Türkozan and Kaska, 2010
Turkey	Tuzla	2006	Türkozan and Kaska, 2010
Turkey	Karatas	1989	Türkozan and Kaska, 2010
Turkey	Agyatan	2006	Türkozan and Kaska, 2010
Turkey	Yelkoma	1996	Türkozan and Kaska, 2010
Turkey	Yumurtalik	2006	Türkozan and Kaska, 2010
DPS 3: SOUTH ATLANTIC			
Brazil	Atol das Rocas	2005-2008	Bellini *et al.*, 2013
Brazil	Isla Trindade	2008-2010	Almeida *et al.*, 2011; Projecto Tamar, 2011
Brazil	Fernando de Noronha	2008-2010	Projecto Tamar, 2011

Equatorial Guinea	Bioko	1996/1997-2004/2005 (except 1998/1999 and 1999/2000)	Rader *et al.*, 2006
Guinea Bissau	Joao Vieira (in the Bijagos Archipelago)	2011	da Silva Ferreira, 2012
Guinea Bissau	Orango National Park	1992-1994	Barbosa *et al.*, 1998; Catry *et al.*, 2009
Guinea Bissau	Poilão	2007	Catry *et al.*, 2009
São Tomé and Príncipe	Praia Grande	2007/2008 and 2009/2010	Loureiro *et al.*, 2011
São Tomé and Príncipe	Principe	2009	Loureiro et al., 2011
Suriname	Galibi Reserve	2008-2010	A. Turney, WWF-Suriname, pers. comm, 2012
Suriname	Matapica Reserve	2008-2010	A. Turney, WWF-Suriname, pers. comm, 2012
United Kingdom	Ascension Island	2010-2012	S. Weber, Ascension Island Government, pers. comm., 2013
USA, USVI	Buck Island	2006-2007	Dow *et al.*, 2007
Venezuela	Aves Island	2010	Prieto *et al.*, 2012
DPS 4: SOUTHWEST INDIAN			
Republic of Seychelles	Aldabra	2004-2008	Mortimer *et al.*, 2011; Mortimer, 2012; J. Mortimer unpubl. data.
Republic of Seychelles	Assumption, Cosmoledo, Astove, Farquhar	1981-1983; other years not provided	Mortimer, 1984; J.A. Mortimer , Seychelles DNR, pers. comm., 2013
Republic of Seychelles	Amirantes Group	1981-1983; other years not provided	Mortimer, 1984; J.A. Mortimer , Seychelles DNR, pers. comm., 2013
Republic of Seychelles	Inner Islands	1981-1983; other years not provided	Mortimer, 1984; J.A. Mortimer , Seychelles DNR, pers. comm., 2013
Comoros Islands	Mohéli	2000-2007	Bourjea, 2012
Comoros Islands, France	Mayotte	1998-2006	Bourjea *et al.*, 2007a; Bourjea, 2012
Eparses Islands, France	Tromelin	1987-2008	Lauret-Stepler *et al.*, 2007; Bourjea, 2012
Eparses Islands, France	Europa	1983-2008	Lauret-Stepler *et al.*, 2007; Bourjea, 2012
Eparses Islands, France	Glorieuses	1987-2008	Lauret-Stepler *et al.*, 2007; Bourjea, 2012
Kenya		1997-2000	Okemwa *et al.*, 2004
Madagascar	Nosy Iranja Kely	2003	Bourjea *et al.*, 2006
Madagascar		No survey year provided	Bourjea, 2012
Mauritius - main island		No survey year provided	Bourjea, 2012
Mozambique		2004-2007 and 2010/2011	Costa *et al.*, 2007; Videira *et al.*, 2011; Bourjea, 2012; Garnier *et al.*,

			2012
Tanzania, including Zanzibar		No survey year provided	Muir, 2005; Bourjea, 2012
United Kingdom (administered) but claimed by Mauritius	Chagos Archipelago	1996	Mortimer and Day, 1999
DPS 5: NORTH INDIAN			
Djibouti		2004	PERSGA/GEF, 2004
Egypt	Egypt (Wadi Al-Gimal, Ras Banas, Sarenka, Siyal, Zabargad, and Rowabill Island)	2012	PERSGA/GEF, 2004
Egypt	Ras Bagdadi	2001-2006	Hanafy, 2012
Egypt	Ras Honkorab	2011	Mancini, 2012
Egypt	Ras Shartib	1967 and 1969	Groombridge and Luxmoore, 1989
Egypt	Sharm El-Sheikh	2011	Mancini, 2012
Egypt	Umm Al-abass	2001-2007	Hanafy, 2012
Egypt	Zabargard Island	2001-2012	Hanafy, 2012; El-Sadek *et al.*, 2013
Iran	Iran	1982	Ross and Barwani, 1982
Kuwait	Qaru	2008 and 2011	Rees *et al.*, 2013
Kuwait	Umm Al-Maradim	2008	Papathanasopoulou, 2010
Oman	Oman (total)	1977-1986 and 1990	Ross and Barwani, 1982; Grobler *et al.*, 2001
Oman	Al Halaniyat Islands	1979	Salm *et al.*, 1993
Oman	Batinah	1990	PERSGA/GEF, 2004
Oman	Daymaniyat Islands	1986	Salm *et al.*, 1993
Oman	Hasik to Ra's Hasik	No survey year provided	Salm *et al.*, 1993
Oman	Masirah Island	1977-1986	Grobler *et al.*, 2001
Oman	Musandam Island	1990	Salm *et al.*, 1993
Oman	North coast of Ras Al-Hadd	1989	Salm *et al.*, 1993
Oman	Ras Al-Hadd	1978; 2006-2007	Ross, 1979; AlKindi *et al.*, 2008
Oman	Ra's Jifan to Ra's Jibsh	2000	Salm *et al.*, 1993
Oman	Ra's Madrakah area	No survey year provided	Salm *et al.*, 1993
Oman	Ra's Nuss	No survey year provided	Salm *et al.*, 1993
Oman	South of Hadbin	No survey year provided	Salm *et al.*, 1993
Oman	Sharbithat area	No survey year provided	Salm *et al.*, 1993
Yemen	Sharma	1966, 1972, and 1999	Groombridge and Luxmoore, 1989; Seminoff *et al.*, 2007
Yemen	Ras Sharma	No survey year provided	PERSGA/GEF, 2004
Pakistan	Gwadar and Pasni	1997	Groombridge *et al.*, 1988
Pakistan	Hawkes Bay and Sandpit	1980-1997	Firdous, 2001

Pakistan	Daran Beach, Jiwani	1999-2008	Waqas *et al.*, 2011
Pakistan	Kamgar Beach at Ormara	1986-1987	Groombridge *et al.*, 1988
Somalia		No survey year provided	PERSGA/GEF, 2004
Sudan		No survey year provided	PERSGA/GEF, 2004
Saudi Arabia	Karan and Jana Island	1986- 1997 and 1991-1992	Al-Merghani *et al.*, 2000; Pilcher, 2000
Saudi Arabia	Juraid Island	1991	Pilcher, 2000
Saudi Arabia	Ras Baridi	1987-1995 and 2003	Al-Merghani *et al.*, 2000; Pilcher and Al-Merghani, 2000; PERSGA/GEF, 2004
Saudi Arabia	Saudi Arabia	No survey year provided	Ross and Barwani, 1982
India	Gujarat	1999-2000 and 2004-2005	Sunderraj *et al.*, 2006a; 2006b; K. Shanker, Indian Institute of Science, pers. comm., 2013
India	Suheli Island	2012	K. Shanker, Indian Institute of Science, pers. comm., 2013
Sri Lanka	Rewaka Beach	1996-2000 and 2009-1010	Kapurusinghe, 2006; Ekanayake *et al.*, 2011
Sri Lanka	Kosgoda	2003-2008	Ekanayake *et al.*, 2010
United Arab Emirates	Sir Bu Nair Island	2012	Al Suweidi *et al.*, 2012
DPS 6: EAST INDIAN/WEST PACIFIC			
Northern Australia	Wellesley Group (includes Bountiful Island, Pisonia and Rocky Islands near Mornington Island) 3 sites	No survey year provided	EPA Queensland Turtle Conservation Project unpublished data CITED in Limpus, 2009
Northern Australia	NOTE: Eastern Arnhem Land, Groote Eylandt and Sir Edward Pellew Islands (3 sites)	1991-2004	Chatto and Baker, 2008
Northern Australia	Northern Territory: Arnhem Wessel, Cobourg, Groote, Groote Eyelant, Pellew, Tiwi, Cobourg, and Cobourg Peninsula (new sites differ from estimate above)	1991-2004	Chatto and Baker, 2008
Australia	Ashmore Reef	1994-1998	Whiting *et al.*, 2000; Jensen, 2010
Australia	Barrow Island	1998-2005	Pendoley, 2005; Jensen, 2010
Australia	Bigge Island	1999	RPS Environmental Pty Ltd., 2008
Australia	Cartier Island	1998	Whiting *et al.*, 2000
Australia	Cassini Island	1999	RPS Environmental Pty Ltd., 2008
Australia	Cape Range NP	2008-2010	Bool *et al.*, 2009; Gourlay *et al.*, 2010
Australia	Cocos (Keeling) Islands	1999	Limpus *et al.*, 2003; Harvey *et al.*, 2005; Limpus, 2009
Australia	Coral Bay	2008-2009	Bool *et al.*, 2009
Australia	Hat Point	1999	RPS Environmental Pty Ltd., 2008
Australia	Jane Bay	2001-2002	RPS Environmental Pty Ltd., 2008

Australia	Jurabi Coastal Park, Ningaloo MP	2001 and 2002	Waayers, 2003; "Australian Government Department of Sustainability, Environment, Water, Population and Communities Species Profile and Threats Database. *Chelonia mydas* - green turtle," 2012
Australia	Lacepedes Islands	2006	RPS Environmental Pty Ltd., 2008
Australia	Lamarck Island	1999	Limpus *et al.*, 2003; Harvey *et al.*, 2005; Limpus, 2009
Australia	Lowendal Island	1998-2005	Pendoley, 2005
Australia	Maret Islands	2006 and 2007	RPS Environmental Pty Ltd., 2008
Australia	Montalivet Island	2006	RPS Environmental Pty Ltd., 2008
Australia	Montebello Island	1998-2005	Pendoley, 2005
Australia	Muiron Islands, Ningaloo MP	1999	"Australian Government Department of Sustainability, Environment, Water, Population and Communities Species Profile and Threats Database. *Chelonia mydas* - green turtle," 2012
Australia	Ningaloo, North West Cape	1988-1989, 199/2000 and 2008-2011	Prince, 2003; Markovina, 2008; Bool *et al.*, 2009; Gourlay *et al.*, 2010; Kelliher *et al.*, 2011
Australia	Red Bluff	1999	Limpus *et al.*, 2003; Harvey *et al.*, 2005; Limpus, 2009
Brunei	Brunei	1999-2008	"National turtle management and conservation in Brunei Darussalam," 2010
India	Great Nicobar Island	1991	Tiwari, 2012
India	Little Nicobar Island	1991	Tiwari, 2012
India	Andaman and Nicobar Islands	2001	Bhaskar, 1979, 1984, 1993; Andrews *et al.*, 2001, 2006a, 2006b; Namboothri *et al.*, 2012
Bangladesh	St. Martin Island	1996 to 2001	Islam, 2002
Indonesia	Belambangan Island	2000	Dermawan, 2002
Indonesia	Berau	1940-1949, 1984, 1985-2000, and 2003-2009	Schulz, 1984; Dermawan, 2002; Adnyana *et al.*, 2008; Reischig *et al.*, 2012
Indonesia	Bilang-Bilangan	2008–2009	Reischig et al., 2012
Indonesia	Enu	1997–1998	Dethmers, 2010
Indonesia	Maluku	1997-1998	Dethmers, 2010
Indonesia	Java	2010	Muhara and Herlina, 2012
Indonesia	East Java	1991-1995	Muhara and Herlina, 2012
Indonesia	Mataha	2008	Reischig et al., 2012
Indonesia	Pangumbahan	2010	Muhara and Herlina, 201
Indonesia	Sangalaki	2003–2009	Reischig et al., 2012
Japan	east Ishigaki Island, Yeayama Islands, Okinawa	1995-2003	Abe *et al.*, 2003
Japan	Okinawa	1995-1996	Kikukawa and Kamezaki, 1999

Malaysia	Tioman Island/Peninsula Malaysia	2011	Fisher, 2012
Malaysia	Redang Island, Terengganu	1993-2008	Chan, 2010
Malaysia	Sabah Turtle Island Park (Gulisaan Island, Bakkunaan Kechil, Selingaan Island)	1966-2011	de Silva, 1982; Basintal, 2002; P. Basintal, Saba Parks Sept., Malaysia, pers. comm., 2011
Malaysia	Sarawak Turtle Island	1933-1934, 1936, 1948-2001	Groombridge and Luxmoore, 1989; Chan, 2006
Malaysia	Terengganu, Peninsula Malaysia	1984-2000	Liew, 2002
Myanmar	Kaingthaung Kyun	1999	Thorbjarnarson *et al.*, 2000
Myanmar	Thameehla Island	1986-2007	Lwin, 2009a
Philippines	Turtle Islands, Tawi-Tawi	1984-2000	Cruz, 2002
Philippines	Baguan Island	2008-2012	Pawikan Conservation Project, 2013
Philippines	Taganak	2008-2012	Pawikan Conservation Project, 2013
Philippines	Lihiman	2008-2012	Pawikan Conservation Project, 2013
Philippines	Langaan	2008-2012	Pawikan Conservation Project, 2013
Philippines	Great Bakkungaan	2008-2012	Pawikan Conservation Project, 2013
Taiwan	Lanyu	1997-2011	Cheng *et al.*, 2009; King *et al.*, 2013
Taiwan	LiuChiu Island	2011	King *et al.*, 2013
Taiwan	Taipin Tao	1995	Cheng, 1996
Taiwan	Wan-an	2010-2011	Cheng *et al.*, 2008; Chen *et al.*, 2010; King *et al.*, 2013
Thailand	Huyong Island	2004	Yasuda *et al.*, 2006
Thailand	Khram, Ira and Chan	2011	Charuchinda and Monanunsap, 1998; Charuchinda *et al.*, 2002; Hykle, 2012
Thailand	Tarutao National Park	1993/1994	Settle, 1995
Viet Nam	Con Dao Island	2001	Hien, 2002; Hamann *et al.*, 2006a
Viet Nam	Gulf of Thailand	1965	Hamann *et al.*, 2006a
Viet Nam	Nui Chua Nature Reserve	2001-2004	Hamann *et al.*, 2005
DPS 7: CENTRAL WEST PACIFIC			
Commonwealth of the Northern Mariana Islands	Rota	2012	Kolinski *et al.*, 2006; Palacios, 2012b
Commonwealth of the Northern Mariana Islands	Tinian	2012	Kolinski *et al.*, 2004; Palacios, 2012b; Wenninger, 2012
Commonwealth of the Northern Mariana Islands	Saipan	2012	Kolinski *et al.*, 2001; Palacios, 2012b
Fedrated States of Micronesia	Fanang	No survey year provided	Maison *et al.*, 2010
Fedrated States of Micronesia	Gaferut	No survey year provided	J. Cruce, Ocean Society, pers. comm., 2013
Fedrated States of Micronesia	Oroluk Atoll	1990	Naughton, 1991
Fedrated States of Micronesia	Pikelot	1970	Pritchard, 1995

Fedrated States of Micronesia	Sorol Atoll	No survey year provided	J. Cruce, Ocean Society, pers. comm., 2013
Fedrated States of Micronesia	Murilo Atoll	1993	Maison *et al.*, 2010
Fedrated States of Micronesia	East Fayu	1993	Maison *et al.*, 2010
Fedrated States of Micronesia	Olimarao Atoll	1990	Hachiglou *et al.*, 1991
Fedrated States of Micronesia	Elato Atoll	1992	Kolinski, 1994
Fedrated States of Micronesia	Ngulu Atoll	1992	Kolinski, 1994
Fedrated States of Micronesia	Ulithi Atoll Loosiep Island	2010-2012	J. Cruce, Ocean Society, pers. comm., 2013
Fedrated States of Micronesia	Ulithi Atoll Gielop and Iar Island	2010-2012	J. Cruce, Ocean Society, pers. comm., 2013
Guam	Island of Guam (and Cocos)	2010; 2008-2010	Maison *et al.*, 2010; Guam Division of Aquatic and Wildlife Resources, 2011, 2012
Indonesia	Jamursba-Medi	1995-1997	Dermawan 2002; Hitipeuw and Maturbongs 2002
Japan	Mukojima	2010-2012	H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012
Japan	Hahajima	2010-2012	H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012
Japan	Chichijima	2010-2012	Chaloupka *et al.*, 2008 H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012
Marshall Islands	Ailuk	1997	McCoy, 2004
Marshall Islands	Wotho	1988	Thomas, 1989
Marshall Islands	Wotje Atoll	2003	McCoy, 2004
Marshall Islands	Rongerik Atoll	2003	McCoy, 2004
Marshall Islands	Bikini	1992	Maison *et al.*, 2010
Marshall Islands	Enewetak	1992	Maison *et al.*, 2010
Marshall Islands	Erikub	1992	McCoy, 2004
Marshall Islands	Jemo	1992	Maison *et al.*, 2010
Marshall Islands	Bikar Atoll	1992	NMFS and USFWS, 1998
Palau	Pulo Ana Island	2005	Bureau of Marine Resources, 2005
Palau	Kayangel Atoll	2005	Bureau of Marine Resources, 2005
Palau	Ngarchelong State	2005	Bureau of Marine Resources, 2005
Palau	Ngerechur Island	2005	Palau Bureau of Marine Resources, 2008
Palau	Helen Island	2005	Barr, 2006
Palau	Merir Island, Sonsorol State	2007/2008	Bureau of Marine Resources, 2005; Barr, 2006; Palau Bureau of Marine Resources, 2008
Papau New Guinea	Lemus	2007	National Fisheries Authority Papua New Guinea, 2007
Papau New Guinea	Mussau	2007	National Fisheries Authority Papua New Guinea, 2007
Papau New Guinea	Nago	2007	National Fisheries Authority Papua

			New Guinea, 2007
Papau New Guinea	Nusalaman (Nusalomon)	2007	National Fisheries Authority Papua New Guinea, 2007
Papau New Guinea	Ral	2007	National Fisheries Authority Papua New Guinea, 2007
Papau New Guinea	Usen (Usang)	2007	National Fisheries Authority Papua New Guinea, 2007
Papua New Guinea	Atmago (Egmakau)	2007	National Fisheries Authority Papua New Guinea, 2007
Papua New Guinea	Emirau	2007	National Fisheries Authority Papua New Guinea, 2007
Solomon Islands	Hakelake Island	1995	Maison *et al.*, 2010
Solomon Islands	Kerehikapa Island	1995	Maison *et al.*, 2010
Solomon Islands	Ausilala	1981	Maison *et al.*, 2010
Solomon Islands	Balaka	1981	Maison *et al.*, 2010
Solomon Islands	Maifu	1981	Maison *et al.*, 2010
Solomon Islands	Malaulaul	1981	Maison *et al.*, 2010
Solomon Islands	Malaupaina	1981	Maison *et al.*, 2010
Solomon Islands	Wagina	1981	Maison *et al.*, 2010
DPS 8: SOUTHWEST PACIFIC			
Australia	All rookeries in northern GBR		Limpus, 2009
Australia	Raine Island (M)	2004	Limpus *et al.*, 2003; Chaloupka *et al.*, 2008; Limpus, 2009
Australia	Moulter Cay (M)	1997-2001, 2004	Limpus *et al.*, 2003; Limpus, 2009
Australia	No. 7 Sandbank (M)	1989, 1991-1992	Limpus *et al.*, 2003; Limpus, 2009
Australia	No. 8 Sandbank (M)	1997	Limpus *et al.*, 2003; Limpus, 2009
Australia	Bramble Cay (m)	1976, 1977, 1979, 1980	Limpus *et al.*, 2003; Limpus, 2009
Australia	Other northern GBR (m)	1981-1997	Limpus *et al.*, 2003
Australia	All rookeries in southern GBR		Limpus, 2009
Australia	Heron Island (M)	1999-2004	Chaloupka *et al.*, 2008; Limpus, 2009
Australia	Rest of Capricorn Bunker Group (M)	1998/9-2003/4	Limpus, 2009
Australia	Rest of southern GBR (m)		Limpus, 2009
Australia	All rookeries in Coral Sea		Limpus, 2009
New Caledonia	All rookeries in New Caledonia		Limpus, 2009
New Caledonia	Huon, Leleizour, Fabre in the atolls d'Entrecasteaux in northern New Caledonia	2007-2011	Read and Fonfreyde, 2011
Vanuatu	Bamboo Bay (west coast of Malekula Island)	2006	MacKay and Petro, 2013
DPS 9: CENTRAL SOUTH PACIFIC			
American Samoa	Tutuila	2007-2013	A. Tagarino, unpub. data
American Samoa	Swains Atoll	2007-2013	A. Tagarino, unpub. data

American Samoa	Rose Atoll	2006-2012	K. Van Houtan, unpub. data
Cook Islands	Palmertson Atoll	2010	White, 2012b
Fiji	Nanuku Levu	2006	Batibasaga *et al.*, 2006
Fiji	Nukumbalati	2006	Batibasaga *et al.*, 2006
French Polynesia	Maupiti	2010	Conservation International Pacific Islands Program, 2013
French Polynesia	Maiao	2009	Gouin and Petit, 2010
French Polynesia	Tupai	1995	Conservation International Pacific Islands Program, 2013
French Polynesia	Tikehau	2007-2010	Gouin and Petit, 2010
French Polynesia	Tetiaroa	2010	Petit, 2011
French Polynesia	Bora Bora	2010	Maison *et al.*, 2010
French Polynesia	Motu One	1991	Trevor, 2009
French Polynesia	Mopelia	2010	Goutenègre *et al.*, 2011
French Polynesia	Tuamotus	1930	Emory, 1975
French Polynesia	Scilly Atoll	1991	Balazs *et al.*, 1995
Kiribati	Birnie	2002	Obura and Stone, 2002
Kiribati	McKean	2000	Obura and Stone, 2002
Kiribati	Orona	2002	Obura and Stone, 2002
Kiribati	Caroline	1990s	Teeb'aki, 1992
Kiribati	Phoenix	2002	Obura and Stone, 2002
Kiribati	Teraina (Washington)	1990s	Teeb'aki, 1992
Kiribati	Malden	1990s	Teeb'aki, 1992
Kiribati	Tarawa	2007	Bell *et al.*, 2009
Kiribati	Manra	2002	Obura and Stone, 2002
Kiribati	Kanton	2002	Obura and Stone, 2002
Kiribati	Nikumaroro	2002	Obura and Stone, 2002
Kiribati	Enderbury	2002	Obura and Stone, 2002
Tokelau	Atafu	1970s	Balazs, 1983
Tokelau	Fakaofu	1970s	Balazs, 1983
Tokelau	Nukunonu	1970s	Balazs, 1983
Tonga	Luanamo	2007	Bell *et al.*, 2009
Tonga	Nukulei	2007	Bell *et al.*, 2009
Tonga	Fonuaika	2007	Bell *et al.*, 2009
Tuvalu	Funafuti	2006	Alefaio and Alefaio, 2006
UK Overseas Territory	Henderson, Pitchairn Islands	1991	Brooke, 1995
DPS 10: CENTRAL NORTH PACIFIC			
USA, HI	French Frigate Shoals	2009-2012	Kittinger *et al.*, 2013
USA, HI	Kamehame	2010-2012	PIFSC unpub. data
USA, HI	Laysan Island	2011	Kittinger *et al.*, 2013
USA, HI	Lisianski Island	2011	Kittinger *et al.*, 2013
USA, HI	Midway Atoll	2011	Kittinger *et al.*, 2013

USA, HI	Lanai	2010-2012	PIFSC unpub. data
USA, HI	Kahoolawe	2010-2012	PIFSC unpub. data
USA, HI	Maui	2010-2012	Department of Land and Natural Resources, 2013
USA, HI	Oahu	2010-2012	PIFSC unpub. data
USA, HI	Kauai	2010-2012	PIFSC unpub. data
USA, HI	Pearl Hermes Reef	2011	Kittinger *et al.*, 2013
USA, HI	Molokai	2011	Kittinger *et al.*, 2013
DPS 11: EAST PACIFIC			
Mexico	Colola, Michoacan	2010-12 seasons	Delgado-Trejo and Alvarado-Figueroa, 2012
Mexico	llorona, Michocan	2007 season	Delgado-Trejo and Alvarado-Figueroa, 2012
Mexico	Bahia Maruata, Michoacan	2007 season	Delgado-Trejo and Alvarado-Figueroa, 2012
Mexico	Motin de Oro, Michocan	2007 season	Delgado-Trejo and Alvarado-Figueroa, 2012
Mexico	Arenas Blancas, Michoacan	2007 season	Delgado-Trejo and Alvarado-Figueroa, 2012
Mexico	Cape Region, BCS	2007-09 seasons	Tiburcio-Pinto *et al.*, 2012
Mexico	Revillagigedos, MX	November - December 2008	Holroyd and Telfry, 2010
Guatemala	Hawaii	N/A	J. Seminoff, NMFS, pers. comm., 2013
El Salvador	Sasonate	N/A	J. Seminoff, NMFS, pers. comm., 2013
El Salvador	San Vincente	N/A	J. Seminoff, NMFS, pers. comm., 2013
Costa Rica	Playa Junquillal	June 2012 – March 2013	M. Heidermeyer, U. Costa Rica, pers. Comm. 2013
Costa Rica	Playa San José, Bat Islands	November - March 2013	L. Fonseca, unpubl. data
Costa Rica	Playa Coloradas	4 observations, each 2 in January, March 2013	M. Heidermeyer, U. Costa Rica, pers. comm. 2013
Costa Rica	Playa Nancite	2012	Fonseca *et al.*, 2011
Costa Rica	Playa Naranjo	2012	L. Fonseca, WIDECAST, pers. comm., 2012; M. Heidermeyer, U. Costa Rica, pers. comm., 2014.
Costa Rica	Playa Cabuyal	2012	P. Santidrián-Tomillo, Leatherback Trust, pers. comm., 2013
Costa Rica	Playa Prieta	1 time observation January 2010	Blanco and Santidrián, 2011
Costa Rica	Playa Virador	1 time observation January 2010	Blanco and Santidrián, 2011
Costa Rica	Playa Matapalo	1 time observation January 2010	Blanco and Santidrián, 2011
Costa Rica	Playa Blanca	1 time observation	Blanco and Santidrián, 2011

		January 2010	
Costa Rica	Nombre de Jesús-Zapotillal	2012-2014	E. Vélez-Carballo, Asociación Kuemar, and R. Piedra-Chacón, Minae-SINAC, unpubl. data.
Costa Rica	Playa Grande-Ventanas	Year-round	Solano Martinez, 2013
Costa Rica	Playa Langosta	November 2012 - March 2013	Piedra Chacón, MINAE, pers. comm., 2013
Costa Rica	Playa Avellanas	October 2011 - March 2012	Ward *et al.*, 2012
Costa Rica	Playa Lagartillo	October 2011 - March 2012	Ward *et al.*, 2012
Costa Rica	Playa Callejones	October 2011 - March 2012	Ward *et al.*, 2012
Costa Rica	Playa Blanca	October 2011 - March 2012	Ward *et al.*, 2012
Costa Rica	Playa Junquillal	Year-round	Francia, 2011
Costa Rica	Playa Ostional	October 2011 - April 2012	Quirós Pereira and Figgener, 2011
Costa Rica	Playa Buena Vista	July - December 2009	Salano Cordero *et al.*, 2010
Costa Rica	Playa Camaronal	2012	Solano Martinez, 2013
Costa Rica	Playa Corozalito	June - December 2012	Viejobueno *et al.*, 2013
Costa Rica	Playa San Miguel	2012	S. Viejobueno, pers. comm., 2013
Costa Rica	Playa Caletas	June - December 2012	Viejobueno *et al.*, 2013
Costa Rica	Punta Banco	June - December 2010	Viejobueno *et al.*, 2013
Colombia	El Valle	N/A	J. Seminoff, NMFS, pers. comm., 2013
Colombia	Isla Gorgona	2007-09 seasons	Amorocho and Reina, 2008; Amorocho *et al.*, 2012
Ecuador	Galapagos (4 beaches)	2003-05 seasons	Zárate *et al.*, 2006; P. Zárate, U. Florida, Unpubl. Data
Ecuador	Ecuador Mainland	2010	Pena Mosquera *et al.*, 2009

Appendix 3. Summary of data sources for the green turtle nesting trend information presented in the Report.

Nesting Site Name	Country	Units	Seasons of data	Range of Years	Reference
DPS 1: NORTH ATLANTIC					
Tortuguero	Costa Rica	AF	41	1971-2011	Harrison and Troëng, 2005; de Haro and Troëng, 2006; de Haro and Harrison, 2007; Chaloupka *et al.*, 2008; Debade *et al.*, 2008; Nolasco del Aguila *et al.*, 2009; Atkinson *et al.*, 2010, 2011; Gonzalez Prieto and Harrison, 2012
Core Index Beaches, Florida	United States of America	AF	24	1989-2012	Florida Fish and Wildlife Conservation Commission, 2013; D. Bagley, Univ. Central FL, pers. comm., 2013
Isla Aguada, Campeche	Mexico	AF	21	1992-2012	Guzmán-Hernández, 2000, 2001, 2002, 2003, 2005, 2006b, 2006c; Guzmán-Hernández *et al.*, 2008; Guzmán-Hernández and García Alvarado, 2013a, 2013b, 2009, 2010, 2011
Guanahacabibes Peninsula	Cuba	AF	15	1998-2012	Azanza Ricardo, 2009; Azanza Ricardo *et al.*, 2013b
El Cuyo, Yucatán	Mexico	AN	14	1990-2000, 2002-2004	K. Lopez, pers. comm. in Seminoff *et al.,* 2004; Xavier *et al.*, 2006
Guanal	Cuba	AN	14	1998-2011	Nodarse *et al.*, 2010; Azanza *et al.*, 2013
San Felipe	Cuba	AN	11	2001-2011	Nodarse *et al.*, 2010; Azanza *et al.*, 2013
DPS 2: MEDITERRANEAN					
West Coast	Cyprus	AF	20	1989-2008	Demetropoulos and Hadjichristophorou, 2009, 2010
Akrotiri	Cyprus	AN	17	1994-1997, 1999-2011	Charilaou and Perdiou, 2012
Israel	Israel	AN	31	1985-2011 (some years have no nests)	Y. Levy, Israel Sea Turtle Rescue Centre, pers. comm., 2012

DPS 2: MEDITERRANEAN Continued					
Akyatan	Turkey	AN	17	1988, 1991-1992, 1994-1998, 2000-2001, 2006-2011	Brown and MacDonald, 1995; Demirayak, 1999; Kasparek *et al.*, 2001; Türkozan *et al.*, 2007; Yilmaz *et al.*, 2009; Yılmaz *et al.,* in prep
Kazanli	Turkey	AN	13	1988, 1990, 1993-1994, 1996, 2000-2007	Yerli and Canbolat, 1998; Demirayak, 1999; Aureggi, 2001; Kasparek *et al.*, 2001; Elmaz, 2005; Venizelos and Kasparek, 2006; Kasparek, 2007; Venizelos, 2007; Kaska, 2008; Casale and Margaritoulis, 2010
Samandag	Turkey	AN	11	1988, 1994, 1996, 1999, 2001-2007	Yerli and Canbolat, 1998; Kasparek *et al.*, 2001; Yalçın-Özdilek, 2007; Kaska, 2008
DPS 3: SOUTH ATLANTIC					
Ascension Island	United Kingdom	AF	13	1822, 1977-1978, 1981-1982, 1990, 1992, 1999-2004	Mortimer and Carr, 1987; Lahanas *et al.*, 1998; Hays *et al.*, 2002; Broderick *et al.*, 2006; Pintus *et al.*, 2009
Atol das Rocas	Brazil	AN	18	1993-1997, 2001-2006, 2008	Bellini *et al.*, 2013
Galibi Reserve and Matapica	Suriname	AN	33	1968-1989, 2000-2010	Schulz, 1975; Reichart and Fretey, 1993; Godfrey *et al.*, 1996; Hilterman *et al.*, 2001; A. Turny, pers. comm., 2012
DPS 4: SOUTHWEST INDIAN					
Europa, Esparses Islands	France	AT	23	1983-2005	Lauret-Stepler *et al.*, 2007

Tromelin, Esparses Islands	France	AT	20	1986-2005	Lauret-Stepler *et al.*, 2007
Glorieuses, Esparses Islands	France	AT	19	1987-2005	Lauret-Stepler *et al.*, 2007
DPS 5: NORTH INDIAN					
Daran Beach, Jiwani	Pakistan	AN	10	1999-2008	Waqas *et al.*, 2011
Zabargard Island	Egypt	AF	10	2001, 2003-2010, 2012	Hanafy, 2012; El-Sadek *et al.*, 2013
DPS 6: EAST INDIAN/WEST PACIFIC					
Sabah Turtle Islands	Malaysia	AF	32	1979-2010	Basintal, 2002; P. Basintal, Saba Parks Dept., Mayasia, pers. comm., 2011
Thameehla Island	Myanmar	AF	22	1986-2007	Lwin, 2009a
Royal Thai Navy Center	Thailand	AF	20	1992-2011	Hykle, 2012
Redang Island, Terengganu	Malaysia	AF	16	1993-2008	Chan, 2010
Wan-an	Taiwan, Province of China	AF	21	1992-2012	I.-J. Cheng, Taiwan Institue of Marine Biology, pers. comm., 2013; King *et al.*, 2013
Lanyu	Taiwan, Province of China	AF	16	1997-2012	Cheng *et al.*, 2009; I.-J. Cheng, Taiwan Institue of Marine Biology, pers. comm., 2013; King *et al.*, 2013
DPS 7: CENTRAL WEST PACIFIC					
Chichijima, Ogasawara Islands	Japan	AF	35	1978-2012	Chaloupka *et al.*, 2008; H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012
Hahajima, Ogasawara Islands	Japan	AN	23	1988-2008, 2010-2011	H. Suganuma, Everlasting Nature of Asia, pers. comm., 2012

DPS 8: SOUTHWEST PACIFIC					
Heron Island	Australia	AF	31	1974-2004	Chaloupka *et al.*, 2008; Limpus, 2009
Raine Island	Australia	AF	25	1976-1982, 1984-1989, 1991-2001, 2004	Limpus *et al.*, 2007; Chaloupka *et al.*, 2008; Limpus, 2009
DPS 9: CENTRAL SOUTH PACIFIC					
No time series due to inconsistent data collection effort or protocol					
DPS 10: CENTRAL NORTH PACIFIC					
East Island, French Frigate Shoals	United States of America	AF	40	1973-2012	Chaloupka *et al.*, 2008; Wetherall, 2012
Maui	United States of America	AN	12	2000-2004, 2006-2012	Department of Land and Natural Resources, 2013
DPS 11: EAST PACIFIC					
Colola, Michoacan	Mexico	AF	30	1982-2011	Delgado-Trejo and Alvarado-Figueroa, 2012

Appendix 4. Green Turtle Status Review Team risk threshold voting form. This form is modified from its original version to fit the following pages.

The following is a voting framework for the Green Turtle Status Review Team to characterize the risk of exinction of 11 DPSs distributed globally. The framework below is to be completed for each of these DPSs independently. Step 1 is intended to provide transparency as to what Elements were the most/least influencial in the overall extinction risk voting. Results from Step 1 will be summarized and provided in the Status Review Report. Step 2 characterizes the SRT Expert opinion based on knowledge about Elements 1-6 (see row 13, 14 for listing of Elements).

STEP 1A: RANK THE IMPORTANCE OF CRITICAL ASSESSMENT ELEMENTS 1,2,3, and 4

A. This step one ranking is intended to provide a measure of importance each team member has placed on each input element when judging the Population Extinction Risk. Risks for each VTP element are ranked on a scale of 1 (very low risk) to 5 (very high risk).

B. For Elements 1-4, please give a rank based on the following. These four elements are considered the 'baseline' status for the DPS (ie. Reflect the current risk status for the DPS)

Rank	Risk Category	Description
1	*Very Low Risk*	Unlikely that this element contributes significantly to risk of extinction, either by itself or in combination with other factors.
2	*Low Risk*	Unlikely that this element contributes significantly to risk of extinction by itself, but some concern that it may, in combination with other factors.
3	*Moderate Risk*	This element contributes significantly to long-term risk of extinction, but does not in itself constitute a danger of extinction in the near future.
4	*High Risk*	This element contributes significantly to long-term risk of extinction and is likely to contribute to short-term risk of extinction in the foreseeable future.
5	*Very High Risk*	This factor by itself indicates a high risk of extinction in the near future.

Appendix 4. *Continued*

STEP 1B: RANK THE INFLUENCE OF 5-FACTOR THREAT ANALYSIS ON THE THE DPS (ELEMENT 5)

The 5-FACTOR THREAT ANALYSIS text section for each DPS reflects events that may have predictable consequences for Extinction Risk. In some cases, the threats may have dramatic influences on a population in the future, but have occurred too recently to be reflected in the population data within the Critical Assessment Element table. Examples include a recent fishery that has started that has significant bycatch, or recent human developments that threaten the viability of a nesting beach. When scoring the influence of the 5-Factor Analysis, please provide a rank without consideration of the information presented in the Conservation Efforts section for the respective DPS. This category is scored as follows:

Rank	Description
0	The threats that are not reflected in Elements 1-4 are likely to have minimal effects.
-1	The threats that are not reflected in Elements 1-4 are likely to have moderate effects.
-2	The threats that are not reflected in Elements 1-4 are likely to have substantial effects

STEP 1C: RANK THE INFLUENCE OF CONSERVATION EFFORTS ON THE DPS (ELEMENT 6)

The CONSERVATION EFFORTS text section for each DPS reflects events that may have predictable consequences for Extinction Risk. In some cases, the conservation efforts may have dramatic influences on a population in the future, but have occurred too recently to be reflected in the population data within the Critical Assessment Element Table. Examples include a recently implemented bycatch mitigation program that is curbing the incidental capture of turtles in a fishery, or a new nesting beach conservation program that is finally starting to protect a given beach after many years of local egg harvest. When scoring the influence of the Conservation Efforts information, please provide a rank without consideration of the information presented in the 5-Factor Analysis section for the respective DPS. This category is scored as follows:

0	The Conservation Efforts that are not reflected in Elements 1-4 are likely to have minimal effects.
1	The Conservation Efforts that are not reflected in Elements 1-4 are likely to have moderate effects.
2	The Conservation Efforts that are not reflected in Elements 1-4 are likely to have substantial effects

	Element 1	Element 2	Element 3	Element 4	Element 5
	Abundance	Productivity	Spatial Structure	Diversity / Resilience	Analyses
YOUR VOTE (put scores on this line)	x	x	x	x	x

Appendix 4. *Continued*

STEP 2: EXPERT OPINION ON GREEN TURTLE CRITICAL RISK THRESHOLDS (LIKELIHOOD BINS)

1 Vote will be based on one's expert opinion about an individual DPS reaching a critical risk threshold* within 100 years.

2 Assign 100 points to each row by spreading the 100 points across any number of risk categories from Extreme low to Extreme high (your personal 'probability distribution'). The points should reflect your interpretation of VIABLE TURTLE POPULATION TABLE INFORMATION, THE 5-FACTOR THREAT ANALYSIS, AND THE CONSERVATION EFFORTS SECTION. You can put as many of the points in a single risk category so long as the spread of points reflect the amount of uncertainty in the risk threshold bins.

3 Each Team Member will assign points based on the following question: What is your expert opinion about the probability that the DPS will reach a critical risk threshold within 100 years, throughout all or a significant portion of its range?

4 The mean, range and other statistics pertaining to each column will be presented below the column.

* 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.

This is a continuum with less risk on the left (white) and more risk on the right (black)

	<1%	1-5%	6-10%	11-20%	21-50%
YOUR VOTE (put scores on this line)	x	x	x	x	x

Appendix 5. Description of International Instruments that may provide positive benefits for green turtles.

Considering the worldwide distribution of green turtles, virtually every legal instrument that targets or impacts sea turtles is almost certain to cover green turtles. A summary of the main regulatory instruments from throughout the world that relate to green turtle management is provided below. The pros and cons of many of these were recently evaluated by Hykle (2002) and Tiwari (2002), and a summary of these findings is given when appropriate.

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

In 1998, the environmental ministers of Cote d'Ivoire, Ghana, Togo, Benin, Nigeria, and Cameroon signed the Accra Declaration to strengthen regional capacity to prevent and correct pollution in the LME and prevent and correct degradation of critical habitats. The ministers identified the living resources and management problems in the area. The countries decided on a detailed survey of industries, defined regional effluent standards, instituted community based mangrove restoration activities, and created a campaign for the reduction, recovery, recycling, and re-use of industrial wastes. In 2006, the Guinea Current LME Project expanded the project scope to 10 neighboring countries (Guinea-Bissau, Guinea, Sierra Leone, Liberia, Sao Tome and Principe, Equatorial Guinea, Gabon, Congo Brazzaville, Congo-Kinshasa, and Angola).

African Convention on the Conservation of Nature and Natural Resources (Algiers Convention)

Adopted in September 1968, the contracted states were "to undertake to adopt the measures necessary to ensure conservation, utilization and development of soil, water, floral and faunal resources in accordance with scientific principles and with due regard to the best interests of the people". It was followed by the 1972 Stockholm Conference on the Human Environment and led to the establishment of environmental ministries in African nations and the creation of the United Nations Environment Programme (UNEP) headquartered in Nairobi. The Algiers Convention recently has undergone revision (not yet in force) and its objectives are to enhance environmental protection, foster conservation and sustainable use of natural resources, and harmonize and coordinate policies in these fields with a view to achieving ecologically rational, economically sound, and socially acceptable development policies and programs. Additional information is available at http://www.unep.ch/regionalseas/legal/afr.htm.

Convention on the Conservation of Migratory Species of Wild Animals (CMS)

This Convention, also known as the Bonn Convention or CMS, is an international treaty that focuses on the conservation of migratory species and their habitats. As of January 2013, the Convention had 118 Parties, including Parties from Africa, Central and South America, Asia, Europe, and Oceania. While the Convention has successfully brought together about half the countries of the world with a direct interest in sea turtles, it has yet to realize its full potential

(Hykle, 2002). Its membership does not include a number of key countries, including Brazil, Canada, China, Indonesia, Japan, Mexico, Oman, and the United States. Additional information is available at http://www.cms.int.

Convention on Biological Diversity (CBD)

The primary objectives of this international treaty are: (1) the conservation of biological diversity, 2) the sustainable use of its components, and 3) the fair and equitable sharing of the benefits arising out of the utilization of genetic resources. This Convention has been in force since 1993 and had 193 Parties as of March 2013. While the Convention provides a framework within which broad conservation objectives may be pursued, it does not specifically address sea turtle conservation (Hykle, 2002). Additional information is available at http://www.cbd.int.

Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)

Known as CITES, this Convention was designed to regulate international trade in a wide range of wild animals and plants. CITES was implemented in 1975 and had 173 Parties as of March 2013. Although CITES has been effective at minimizing the international trade of sea turtle products, it does not limit legal harvest within countries, nor does it regulate intra-country commerce of sea turtle products (Hykle, 2002). Additional information is available at http://www.cites.org.

Convention on the Conservation of European Wildlife and Natural Habitats

Also known as the Bern Convention, the goals of this instrument are to conserve wild flora and fauna and their natural habitats, especially those species and habitats whose conservation requires the cooperation of several States, and to promote such cooperation. The Convention was enacted in 1982 and currently includes 51 European and African States and the European Union. Additional information is available at http://www.coe.int/t/dg4/cultureheritage/nature/bern/default_en.asp.

Convention for the Co-operation in the Protection and Development of the Marine and Coastal Environment of the West and Central African Region (Abidjan Convention)

The Abidjan Convention covers the marine environment, coastal zones, and related inland waters from Mauritania to Namibia. The Abidjan Convention countries are Angola, Benin, Cameroon, Cape Verde, Congo, Cote d'Ivoire, Democratic Republic of Congo, Equatorial Guinea, Gabon, Gambia, Ghana, Guinea, Guinea-Bissau, Liberia, Mauritania, Namibia, Nigeria, Sao Tome and Principe, Senegal, Sierra Leone, and Togo. The Abidjan Convention is an agreement for the protection and management of the marine and coastal areas that highlights sources of pollution, including pollution from ships, dumping, land-based sources, exploration and exploitation of the sea-bed, and pollution from or through the atmosphere. The Convention also identifies where co-operative environmental management efforts are needed. These areas of concern include coastal erosion, specially protected areas, combating pollution in cases of emergency, and environmental impact assessment. The Action Plan and the Abidjan Convention were adopted by the Governments in 1981; the Convention entered into force in 1984. Western Sahara and Morocco are not signatories of the Abidjan Convention.

Convention for the Protection of the Marine Environment of the North-East Atlantic

Also called the OSPAR Convention, this 1992 instrument combines and updates the 1972 Oslo Convention against dumping waste in the marine environment and the 1974 Paris Convention addressing marine pollution stemming from land-based sources. The convention is managed by the OSPAR Commission, which is comprised of representatives from 15 signatory nations (Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, The Netherlands, Norway, Portugal, Spain, Sweden, Switzerland, and United Kingdom), as well as the European Commission, representing the European Community. The mission of the OSPAR Convention "…is to conserve marine ecosystems and safeguard human health in the North-East Atlantic by preventing and eliminating pollution; by protecting the marine environment from the adverse effects of human activities; and by contributing to the sustainable use of the seas." Loggerheads are included on the OSPAR List of Threatened and/or Declining Species and Habitats, which is used by the OSPAR Commission for setting priorities for work on the conservation and protection of marine biodiversity. Additional information is available at http://www.ospar.org.

Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region, Specially Protected Areas and Wildlife (SPAW)

Also called the Cartagena Convention, this instrument has been in place since 1986 and had 23 Signatory States as of June 2010. Under this Convention, the component that may relate to green turtles is the Protocol Concerning Specially Protected Areas and Wildlife (SPAW) that has been in place since 2000. The goals are to encourage Parties "to take all appropriate measures to protect and preserve rare or fragile ecosystems, as well as the habitat of depleted, threatened or endangered species, in the Convention area." All six sea turtle species in the Wider Caribbean are listed in Annex II of the protocol, which prohibits (a) the taking, possession or killing (including, to the extent possible, the incidental taking, possession or killing) or commercial trade in such species, their eggs, parts or products, and (b) to the extent possible, the disturbance of such species, particularly during breeding, incubation, estivation, migration, and other periods of biological stress. The SPAW protocol has partnered with WIDECAST to develop a program of work on sea turtle conservation, which has helped many of the Caribbean nations to identify and prioritize their conservation actions through Sea Turtle Recovery Action Plans. Hykle (2002) believes that in view of the limited participation of Caribbean States in the aforementioned Convention on the Conservation of Migratory Species of Wild Animals, the provisions of the SPAW Protocol provide the legal support for domestic conservation measures that might otherwise not have been afforded. Additional information is available at http://www.cep.unep.org/about-cep/spaw.

Convention for the Protection of the Marine Environment and Coastal Area of the South-East Pacific (Lima Convention)

This Convention's signatories include all countries along the Pacific Rim of South America from Panama to Chile. Among other resource management components, this Convention established a protocol for the conservation and management of protected marine resources. Stemming from this Convention is the Commision Permanente del Pacifico Sur (CPPS) that has developed a Marine Turtle Action Plan for the Southeast Pacific that outlines a strategy for protecting and recovering marine turtles in this region.

Convention for the Protection of the Natural Resources and Environment of the South Pacific Region

This Convention, also known as the Noumea Convention, has been in force since 1990 and currently includes 26 Parties. The purpose of the Convention is to protect the marine environment and coastal zones of the South-East Pacific within the 200-mile area of maritime sovereignty and jurisdiction of the Parties, and beyond that area, the high seas up to a distance within which pollution of the high seas may affect that area. Additional information is available at http://www.unep.org/regionalseas/programmes/nonunep/pacific/instruments/default.asp.

Convention Concerning the Protection of the World Cultural and Natural Heritage (World Heritage Convention)

The World Heritage Convention was signed in 1972 and, as of November 2007, 185 states were parties to the Convention. The instrument requires parties to take effective and active measures to protect and conserve habitat of threatened species of animals and plants of scientific or aesthetic value. The World Heritage Convention currently includes 31 marine sites, including important marine turtle habitat such as the Belize Barrier Reef Reserve System, Belize. Additional information is available at http://whc.unesco.org/en/conventiontext.

Convention for the Conservation and Management of Highly Migratory Fish Stocks in the Western and Central Pacific Ocean (WCPF Convention)

The convention entered into force on 19 June 2004. The WCPF Convention draws on many of the provisions of the UN Fish Stocks Agreement [UNFSA] while, at the same time, reflecting the special political, socio-economic, geographical and environmental characteristics of the western and central Pacific Ocean (WCPO) region. The WCPFC Convention seeks to address problems in the management of high seas fisheries resulting from unregulated fishing, over-capitalization, excessive fleet capacity, vessel re-flagging to escape controls, insufficiently selective gear, unreliable databases and insufficient multilateral cooperation in respect to conservation and management of highly migratory fish stocks.

Convention for the Prohibition of Fishing with Long Driftnets in the South Pacific

This regional convention, also known as the Wellington Convention, was adopted in 1989 in Wellington, New Zealand, and entered into force in 1991. The objective of the Convention is "to restrict and prohibit the use of drift nets in the South Pacific region in order to conserve marine living resources." Additional information is available at http://www.mfat.govt.nz/Treaties-and-International-Law/01-Treaties-for-which-NZ-is-Depositary/0-Prohibition-of-Fishing.php.

Council Regulation (EC) No. 1239/98 of 8 June 1998 Amending Regulation (EC) No. 894/97 Laying Down Certain Technical Measures for the Conservation of Fishery Measures (Council of the European Union)

This measure banned the use of driftnets by 1 January 2002 for European fleets. Fleets from other nations fishing in international waters can still use driftnets.

Food and Agricultural Organization Technical Consultation on Sea Turtle-Fishery Interactions

While not a true international instrument for conservation, the 2004 Food and Agriculture Organization of the United Nations' (FAO) technical consultation on sea turtle-fishery interactions was groundbreaking in that it solidified the commitment of the lead United Nations agency for fisheries to reduce sea turtle bycatch in marine fisheries operations. Recommendations from the technical consultation were endorsed by the FAO Committee on Fisheries (COFI) and called for the immediate implementation by member nations and Regional Fishery Management Organizations (RFMOs) of guidelines to reduce sea turtle mortality in fishing operations, developed as part of the technical consultation.

Currently, all five of the tuna RFMOs call on their members and cooperating non-members to adhere to the 2009 FAO "Guidelines to Reduce Sea Turtle Mortality in Fishing Operations," which describes all the gears sea turtles could interact with and the latest mitigation options. The Western and Central Pacific Fisheries Commission (http://www.wcpfc.int) has the most protective measures (CMM 2008-03), which follow the FAO guidelines and ensure safe handling of all captured sea turtles. Fisheries deploying purse seines, to the extent practicable, must avoid encircling sea turtles and release entangled turtles from fish aggregating devices. Longline fishermen must carry line cutters and use dehookers to release sea turtles caught on a line. Longliners must either use large circle hooks, whole finfish bait, or mitigation measures approved by the Scientific Committee and the Technical and Compliance Committee.

The Inter-American Tropical Tuna Convention (http://www.iattc.org/) has a sea turtle resolution, which encompasses the elements in the Western and Central Pacific Fisheries Commission, but does not require the use a specific mitigation device or bait type in longline fisheries. The Inter-American Tropical Tuna Convention has also developed a memorandum of understanding with the Inter-American Convention for the Protection and Conservation of Sea Turtles. The International Commission for the Conservation of Atlantic Tunas (http://www.iccat.int) has a recommendation on sea turtles, which calls for implementing the FAO Guidelines for sea turtles, avoiding encirclement of sea turtles by purse seiners, safely handling and releasing sea turtles, and reporting on interactions. The Commission does not have any specific gear requirements in longline fisheries. The International Commission for the Conservation of Atlantic Tunas is currently undertaking an ecological risk assessment to better understand the impact of its fisheries on sea turtle populations. The Indian Ocean Tuna Commission (http://www.iotc.org/) is also in the process of carrying out an ecological risk assessment for sea turtles. Their turtle measures encompass similar elements of the other organizations but do not require the use of certain gear or bait in longline fisheries. Finally, the Commission for the Conservation of

Southern Bluefin Tuna (http://www.ccsbt.org) supports the measures called for in the Western and Central Pacific Fisheries Commission and the Indian Ocean Tuna Commission.

Other international fisheries organizations that may influence green turtle recovery include the Southeast Atlantic Fisheries Organization (http://www.seafo.org) and the North Atlantic Fisheries Organization (http://nafo.int). These organizations regulate trawl fisheries in their respective Convention areas. Given that sea turtles can be incidentally captured in these fisheries, both organizations have sea turtle resolutions calling on their Parties to implement the FAO Guidelines on sea turtles as well as to report data on sea turtle interactions.

Indian Ocean – South-East Asian Marine Turtle Memorandum of Understanding (IOSEA)

Under the auspices of the Convention of Migratory Species, the IOSEA memorandum of understanding provides a mechanism for States of the Indian Ocean and South-East Asian region, as well as other concerned States, to work together to conserve and replenish depleted marine turtle populations. This collaboration is achieved through the collective implementation of an associated Conservation and Management Plan. Currently, there are 33 Signatory States. The United States became a signatory in 2001. The IOSEA has an active sub-regional group for the Western Indian Ocean, which has improved collaboration amongst sea turtle conservationists in the region. Further, the IOSEA website provides reference materials, satellite tracks, on-line reporting of compliance with the Convention, and information on all international mechanisms currently in place for the conservation of sea turtles. Finally, at the 2012 Sixth Signatory of States meeting in Bangkok, Thailand, the Signatory States agreed to procedures to establish a network of sites of importance for sea turtles in the IOSEA region (http://www.ioseaturtles.org).

Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC)

This Convention is the only binding international treaty dedicated exclusively to sea turtles and sets standards for the conservation of these endangered animals and their habitats with a large emphasis on bycatch reduction. The Convention area is the Pacific and the Atlantic waters of the Americas. Currently, there are 15 Parties. The United States became a Party in 1999. The IAC has worked to adopt fisheries bycatch resolutions, carried out workshops on Caribbean sea turtle conservation, and established collaboration with other agreements such as the Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region and the International Commission for the Conservation of Atlantic Tunas. Additional information is available at http://www.iacseaturtle.org.

International Convention for the Prevention of Pollution from Ships (MARPOL)

The MARPOL Convention is a combination of two treaties adopted in 1973 and 1978 to prevent pollution of the marine environment by ships from operational or accidental causes. The 1973 treaty covered pollution by oil, chemicals, harmful substances in packaged form, sewage and garbage. The 1978 MARPOL Protocol was adopted at a Conference on Tanker Safety and Pollution Prevention which included standards for tanker design and operation. The 1978 Protocol incorporated the 1973 Convention as it had not yet been in force and is known as the International Convention for the Prevention of Marine Pollution from Ships, 1973, as modified by the Protocol of 1978 relating thereto (MARPOL 73/78). The 1978 Convention went into force in 1983 (Annexes I and II). The Convention includes regulations aimed at preventing and minimizing accidental and routine operations pollution from ships. Amendments passed since have updated the convention. To date there are six Annexes with Annexes I and II being mandatory for State Parties and the others being voluntary.

Annex I - Regulations for the Prevention of Pollution by Oil
Annex II - Regulations for the Control of Pollution by Noxious Liquid Substances in Bulk
Annex III - Prevention of Pollution by Harmful Substances Carried by Sea in Packaged Form
Annex IV - Prevention of Pollution by Sewage from Ships
Annex V - Prevention of Pollution by Garbage from Ships
Annex VI - Prevention of Air Pollution from Ships

International Union for Conservation of Nature (IUCN)

The IUCN Species Programme assesses the conservation status of species on a global scale. This assessment provides objective, scientific information on the current status of threatened species. "The IUCN Red List of Threatened Species provides taxonomic, conservation status and distribution information on plants and animals that have been globally evaluated using the IUCN Red List Categories and Criteria. This system is designed to determine the relative risk of extinction, and the main purpose of the IUCN Red List is to catalogue and highlight those plants and animals that are facing a higher risk of global extinction (i.e., those listed as Critically Endangered, Endangered and Vulnerable)." Additional information is available at http://www.iucnredlist.org/about.

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

Signed in 1996, this bilateral Memorandum of Agreement paved the way for the Turtle Islands Heritage Protected Area, which protects very important concentrations of nesting green turtles and hawksbills. In 2004, a tri-national regional action plan and marine protected area for marine turtles was established as part of the Sulu Sulawesi Marine Ecoregion. More information on this action plan can be found at http://www.fishdept.sabah.gov.my/ssme.asp.

Memorandum of Understanding on Association of South East Asian Nations (ASEAN) Sea Turtle Conservation and Protection

The objectives of this Memorandum of Understanding, initiated by ASEAN, are to promote the protection, conservation, replenishing, and recovery of sea turtles and their habitats based on the best available scientific evidence, taking into account the environmental, socio-economic and cultural characteristics of the Parties. It currently has nine signatory states in the South East Asian Region. As the technical arm of ASEAN, the Southeast Asia Fisheries Development Center (SEAFDEC) supports the work of this Memorandum of Understanding. Further, the Japanese Trust Fund in collaboration with the Malaysian government is supporting a project on the research and management of sea turtles in foraging habitats in Southeast Asian waters (http://document.seafdec.or.th/projects/2012/seaturtles.php).

Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa (Abidjan Memorandum)

This MOU was concluded under the auspices of the Convention on the Conservation of Migratory Species of Wild Animals (CMS) and became effective in 1999. The MOU area covers 26 Range States along the Atlantic coast of Africa extending approximately 14,000 km from Morocco to South Africa. The goal of this MOU is to improve the conservation status of marine turtles along the Atlantic Coast of Africa. It aims at safeguarding six marine turtle species – including the green turtle – that are estimated to have rapidly declined in numbers during recent years due to excessive exploitation (both direct and incidental) and the degradation of essential habitats. This includes the protection of hatchlings through adults with particular attention paid to the impacts of fishery bycatch and the need to include local communities in the development and implementation of conservation activities. However, despite this agreement, killing of adult turtles and harvesting of eggs remains rampant in many areas along the Atlantic African coast. Additional information is available at http://www.cms.int/species/africa_turtle/AFRICAturtle_bkgd.htm.

Nairobi Convention for the Protection, Management and Development of the Marine and Coastal Environment of the Eastern African Region

The Nairobi Convention was signed in 1985 and came into force in 1996. The Convention covers 10 States, including five island States in the Western Indian Ocean. The Contracting Parties are Comoros, France (La Reunion), Kenya, Madagascar, Mauritius, Mozambique, Seychelles, Somalia, Tanzania and the Republic of South Africa. This instrument "provides a mechanism for regional cooperation, coordination and collaborative actions, and enables the Contracting Parties to harness resources and expertise from a wide range of stakeholders and interest groups towards solving interlinked problems of the coastal and marine environment." Additional information is available at http://www.unep.org/NairobiConvention.

Protocol Concerning Specially Protected Areas and Biological Diversity in the Mediterranean

This Protocol is under the auspices of the Barcelona Convention of 1976 for the Protection of the Mediterranean Sea against Pollution (amended in 1995). The Protocol has been in force since 1999 and includes general provisions to protect sea turtles and their habitats within the Mediterranean Sea. The Protocol requires Parties to protect, preserve, and manage threatened or endangered species, establish protected areas, and coordinate bilateral or multilateral conservation efforts (Hykle, 2002). In the framework of the Barcelona Convention, to which all Mediterranean countries are parties, the Action Plan for the Conservation of Mediterranean Marine Turtles has been in effect since 1989. Additional information is available at http://www.rac-spa.org.

Ramsar Convention on Wetlands

The Convention on Wetlands, signed in Ramsar, Iran, in 1971, is an intergovernmental treaty, which provides the framework for national action and international cooperation for the conservation and wise use of wetlands and their resources. Currently, there are 158 parties to the convention, with 1,752 wetland sites, including important marine turtle habitat such as the Turtle Beaches and Coral Reefs of Tongaland, South Africa. Additional information is available at http://www.ramsar.org.

Secretariat of the Pacific Regional Environment Programme (SPREP)

SPREP's turtle conservation program seeks to improve knowledge about sea turtles in the Pacific through an active tagging program, as well as maintaining a database to collate information about sea turtle tags in the Pacific. SPREP supports capacity building throughout the central and southwest Pacific. SPREP established an action plan for the Pacific Islands (http://www.sprep.org/).

South-East Atlantic Fisheries Organization (SEAFO)

SEAFO manages fisheries activities in the Southeast Atlantic high seas area, excluding tunas and billfish. SEAFO adopted Resolution 01/06, "to Reduce Sea Turtle Mortality in Fishing Operations," in 2006. The Resolution requires Members to: (1) implement the FAO Guidelines; and (2) establish on-board observer programs to collect information on sea turtle interactions in SEAFO-managed fisheries. This Resolution is not legally binding. Additional information is available at http://www.seafo.org.

Torres Strait Treaty of 1978

The Torres Strait Treaty is an agreement between Australia and Papua New Guinea which describes the boundaries between the two countries and how the sea areas may be used. In defining the two main boundaries – the Seabed Jurisdiction Line and the Fisheries Jurisdiction Line – as well as a 'protected zone', the Treaty takes account of traditional activities (including sea and land use—including Indigenous sea turtle harvest—trade, ceremonies and social gatherings) of the indigenous residents of the Torres Strait area.

United Nations Convention on the Law of the Sea (UNCLOS)

To date, 155 countries, including most mainland countries lining the western Pacific, and the European Community have joined in the convention. The United States has signed the treaty, but the Senate has not ratified it. Aside from its provisions defining ocean boundaries, the convention establishes general obligations for safeguarding the marine environment through mandating sustainable fishing practices and protecting freedom of scientific research on the high seas. Additional information is available at http://www.un.org/Depts/los/index.htm.

United Nations Resolution 44/225 on Large-Scale Pelagic Driftnet Fishing

In 1989, the United Nations called, in a unanimous resolution, for the elimination of all high seas driftnets by 1992. Additional information is available at *http://www.un.org/documents/ga/res/44/a44r225.htm*.

United States Magnuson-Stevens Fishery Conservation and Management Act

The recently reauthorized 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 gram 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."

The MSA also has provisions that extend to fishing activities in waters beyond U.S. jurisdiction. Section 610 calls on the U.S. Secretary of Commerce to identify nations with fishing vessels that are engaged or have been engaged in fishing activities in waters beyond any national jurisdiction or in areas beyond the Exclusive Economic Zone of the United States. This section authorizes NMFS to conduct bilateral consultations with such nations to protect living marine resources. If a nation continues to conduct fishing activities that result in bycatch of protected living marine resources, they can be certified to the United States Congress. A result of this certification could be suspension in the trade of fisheries products. Finally, the Act specifically encourages NMFS to conduct international cooperation and assistance with foreign nations that are identified so that bycatch of protected living marine resources can be reduced.