LEATHERBACK SEA TURTLE
(DERMOCHELYS CORIACEA)

5-YEAR REVIEW:
SUMMARY AND EVALUATION

NATIONAL MARINE FISHERIES SERVICE
OFFICE OF PROTECTED RESOURCES
SILVER SPRING, MARYLAND
AND
U.S. FISH AND WILDLIFE SERVICE
SOUTHEAST REGION
JACKSONVILLE ECOLOGICAL SERVICES OFFICE
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U.S. Department of the Interior

U.S. FISH AND WILDLIFE SERVICE

TABLE OF CONTENTS

5-YEAR REVIEW
Leatherback Sea Turtle/*Dermochelys coriacea*

1.0 GENERAL INFORMATION

1.1 Reviewers

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1.2. Methodology used to complete the review

The National Marine Fisheries Service (NMFS) Office of Protected Resources led the 5-year review with input from the U.S. Fish and Wildlife Service (FWS). The draft document was distributed to NMFS regional offices and science centers and FWS regional and field offices for their review and edits, which were incorporated where appropriate. Our information sources include the final rule listing of this species under the Endangered Species Act (ESA); the recovery plans for the U.S. Pacific, and the U.S. Caribbean, Atlantic, and Gulf of Mexico; peer reviewed scientific publications; unpublished field observations by the Services, State, and other experienced biologists; unpublished survey reports; and notes and communications from other qualified biologists. The public notice for this review was published on October 10, 2012, with a 60-day comment period (77 FR 61573). Commenters submitted information on sea turtle bycatch reduction measures in the Hawaii-based longline fishery, community outreach programs in Papua New Guinea, monitoring and conservation programs in Indonesia, fisheries bycatch, entanglement and ingestion of debris, vessel strikes, and impacts from climate change. Comments received were incorporated as appropriate into the 5-year review.

1.3 Background

1.3.1 FR notice citation announcing initiation of this review

October 10, 2012 (77 FR 61573)

1.3.2 Listing history

<u>Original Listing</u>
FR notice: 35 FR 8491
Date listed: June 2, 1970
Entity listed: Species
Classification: Endangered

1.3.3 Associated rulemakings

Critical Habitat Designation: 43 FR 43688, September 26, 1978. The purpose of this rule was to designate terrestrial critical habitat for the leatherback turtle as follows: U.S. Virgin Islands – A strip of land 0.2 miles wide (from mean high tide inland) at Sandy Point Beach on the western end of the island of St. Croix beginning at the southwest cape to the south and running 1.2 miles northwest and then northeast along the western and northern shoreline, and from the southwest cape 0.7 miles east along the southern shoreline.

Critical Habitat Designation: 44 FR 17710, March 23, 1979. Critical habitat was designated for waters adjacent to Sandy Point, St. Croix, U.S. Virgin Islands, up to and inclusive of the waters from the hundred fathom curve shoreward to the level of mean high tide.

Regulations Consolidation Final Rule: 64 FR 14052, March 23, 1999. The purpose of this rule was to make the regulations regarding implementation of the Endangered Species Act of 1973 (ESA) by NMFS for marine species more concise, better organized, and therefore easier for the public to use.

Critical Habitat Designation: 77 FR 4170, January 26, 2012. The purpose of this rule was to designate marine critical habitat for the leatherback turtle as follows: (1) California—(a) the area bounded by Point Sur then north along the shoreline following the line of extreme low water to Point Arena then west to 38°57’14” N./123°44’26”W. then south along the 200 meter isobath to 36°18’46” N./122°4’43”W. then east to point of origin at Point Sur; (b) the nearshore area from Point Arena to Point Arguello and offshore to a line connecting 38°57’14” N./124°18’36”W. and 34°34’32” N./121°39’51”W. along the 3,000 meter isobaths; (2) Oregon/Washington—the area bounded by Cape Blanco, Oregon north along the shoreline following the line of the extreme low water to Cape Flattery, Washington then north to the U.S./Canada boundary then west and south along the line of the U.S. Exclusive Economic Zone to 47°57’38” N./126°22’54”W. then south along a line approximating the 2,000 meter isobath then east to the point of origin at Cape Blanco.

1.3.4 Review history

National Marine Fisheries Service and U.S. Fish and Wildlife Service. 2007. Leatherback Sea Turtle (*Dermochelys coriacea*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Silver Spring, Maryland and U.S. Fish and Wildlife Service Jacksonville, Florida. 79 pages.

> Conclusion: Retain the listing as an endangered species. However, a review and analysis of the species listing relative to the Distinct Population Segment policy was recommended.

Plotkin, P.T. (Editor). 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. National Marine Fisheries Service, Silver Spring, Maryland. 139 pages.

> Conclusion: Retain the listing as an endangered species.

Mager, A.M., Jr. 1985. Five-year status reviews of sea turtles listed under the Endangered Species Act of 1973. U.S. Department of Commerce, NOAA, National Marine Fisheries Service, St. Petersburg, Florida. 90 pages.

Conclusion: Retain the listing as an endangered species

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FWS also conducted 5-year reviews for the leatherback in 1985 (50 FR 29901) and in 1991 (56 FR 56882). In these reviews, the status of many species was simultaneously evaluated with no in-depth assessment of the five factors or threats as they pertain to the individual species. The notices stated that FWS was seeking any new or additional information reflecting the necessity of a change in the status of the species under review. The notices indicated that if significant data were available warranting a change in a species' classification, the Service would propose a rule to modify the species' status.

Conclusions: Retain listing as endangered throughout its range.

1.3.5 Species' recovery priority number at start of review

National Marine Fisheries Service = 1 (this represents a high magnitude of threat, a high recovery potential, and the presence of conflict with economic activities).
U.S. Fish and Wildlife Service (48 FR 43098) = 1 (this represents a monotypic genus with a high degree of threat and a high recovery potential).

1.3.6 Recovery plans

Name of plan: Recovery Plan for Leatherback Turtles (*Dermochelys coriacea*) in the U.S. Caribbean, Atlantic, and Gulf of Mexico (NMFS and FWS 1992)
Date issued: April 6, 1992

Name of plan: Recovery Plan for U.S. Pacific Populations of the Leatherback Turtle (*Dermochelys coriacea*) (NMFS and FWS 1998)
Date issued: January 12, 1998

Dates of previous plans: Original plan date - September 19, 1984

2.0 REVIEW ANALYSIS

2.1 Application of the 1996 Distinct Population Segment (DPS) policy

2.1.1 Is the species under review a vertebrate? Yes.

2.1.2 Is the species under review listed as a DPS? No.

2.1.3 Is there relevant new information for this species regarding the application of the DPS policy?

Yes. In the 2007 5-year review (NMFS and FWS 2007), we noted that although the current listing is valid based on the best available information, we had preliminary information that indicated an analysis and review of the species should be conducted to determine the application of the DPS policy to the leatherback. Since the species' listing, a substantial amount of information has become available on population structure (through genetic studies) and distribution (through telemetry, tagging, and genetic studies). The Services have not yet fully assembled or analyzed this new information; however, at a minimum, these data appear to indicate a possible separation of populations by ocean basins.

2.2 Recovery Criteria

2.2.1 Does the species have a final, approved recovery plan containing objective, measurable criteria?

No. The existing recovery plans are based on population and management units within ocean basins and do not represent the species' listing. The "Recovery Plan for Leatherback Turtles (*Dermochelys coriacea*) in the U.S. Caribbean, Atlantic, and Gulf of Mexico" was signed in 1992 (NMFS and FWS 1992), and the "Recovery Plan for U.S. Pacific Populations of the Leatherback Turtle (*Dermochelys coriacea*)" was signed in 1998 (NMFS and FWS 1998). The recovery criteria, in these plans, do not strictly adhere to all elements of the Services' Interim Recovery Planning Guidance (http://www.nmfs.noaa.gov/pr/pdfs/recovery/guidance.pdf), but may provide a useful benchmark for measuring progress toward recovery. Thus, we consider progress towards recovery objectives in this section.

Recovery Objectives as written in the U.S. Caribbean, Atlantic, and Gulf of Mexico Recovery Plan

The U.S. populations of leatherback turtles may be considered for delisting if the following conditions are met:

1. The adult female population increases over the next 25 years, as evidenced by a statistically significant trend in the number of nests at Culebra, Puerto Rico, St. Croix, U.S. Virgin Islands, and along the east coast of Florida.

   Status: In Puerto Rico, the adult female population, as evidenced by number of nests reported between 1978 and 2005, increased about 10% annually. However since 2004, nesting has steadily declined in Culebra, which may reflect a shift in nest site fidelity rather than a decline in the female population. In the U.S. Virgin Islands, St. Croix (Sandy Point National Wildlife Refuge), leatherback nesting was estimated to increase at 13% per year from 1994 through 2001. However, nesting data from 2001 through 2010 indicate population growth has slowed, possibly due to fewer new recruits and lowered reproductive output. In Florida, the number of nests increased by 10.2% (range 3.1%-16.3%) annually from 1979 through 2008. For further detail see Section 2.3.1 Biology and Habitat—Abundance and Population Trends.

2. Nesting habitat encompassing at least 75 percent of nesting activity in USVI, Puerto Rico, and Florida is in public ownership.

   Status: Several properties are in Federal ownership as National Wildlife Refuges in Florida (Archie Carr and Hobe Sound), Puerto Rico (Culebra and Vieques), and the U.S. Virgin Islands (Sandy Point National Wildlife Refuge). The extent of nesting activity occurring on properties in protected ownership has not yet been assessed.

3. All priority one tasks have been successfully implemented.

   - <u>Identify and ensure long-term protection of important nesting beaches (Task 113)</u>.

   Status: This task is ongoing. Monitoring and protection programs have been ongoing for over 30 years at Archie Carr and Hobe Sound National Wildlife Refuges in Florida, Culebra and Vieques in Puerto Rico, and Sandy Point National Wildlife Refuge, U.S. Virgin Islands. Habitat conservation plans have been developed for residential development at Tortola Beach and Hamacao, Puerto Rico, and Indian River, St. Johns, and Volusia Counties, Florida.

   - <u>Identify important foraging and other marine habitats and ensure long-term protection (Task 121)</u>.

   Status: This task is ongoing. Research and monitoring have been conducted in Canada on one of the largest seasonal foraging populations of leatherbacks in the Atlantic Ocean. In cooperation with Canada, threats to leatherback turtles in Canadian waters have been identified and addressed, and recovery plans for leatherbacks in Canada have been developed. Research on foraging areas off Massachusetts is ongoing. Juvenile foraging habitat was characterized for Sao Tome and Principe (an island in the Gulf of Guinea off the west coast of Africa).

   - <u>Monitor nesting activity trends on important nesting beaches with standardized surveys (Task 211)</u>.

   Status: This task is ongoing. Long-term monitoring and protection programs have been ongoing in Florida (Archie Carr, Hobe Sound, and Merritt Island National Wildlife Refuges), Puerto Rico (Culebra and Vieques), and U.S. Virgin Islands (Sandy Point National Wildlife Refuge). A preliminary survey of Angola’s coast was conducted to identify nesting areas. For further detail see Section 2.3.1 Biology and Habitat—Abundance and Population Trends.

   - <u>Evaluate nest success and implement appropriate nest protection measures (Task 212)</u>.

   Status: This task is ongoing. Nest monitoring and protection efforts are ongoing at several National Wildlife Refuges in Florida (Archie Carr, Hobe Sound, and Merritt Island), Puerto Rico (Culebra and Vieques), and the U.S. Virgin Islands (Sandy Point

National Wildlife Refuge), as well as on other beaches throughout the species U.S. nesting range.

- Implement measures to reduce capture and mortality from commercial shrimping vessels (Task 2221).

Status: This task is ongoing and is completed for some sectors. In the southeastern U.S., turtle excluder devices are required in most shrimp trawlers, and modifications to improve turtle exclusion have been codified. Efforts are ongoing to provide turtle excluder device outreach and training for various foreign governments. The Sea Turtle Disentanglement Network was established in 2002 to address sea turtle entanglement in the vertical lines of pot and other fishing gear operating in waters off the northeast coast of the United States. Many leatherbacks have been rescued from fishing gear since the Network's inception. For further detail see Section 2.3.2.5 Other natural or manmade factors affecting its continued existence.

- Evaluate the extent of entanglement and ingestion of persistent marine debris (Tasks 2241 and 2242) and formulate and implement appropriate measures to reduce or eliminate persistent marine debris in the marine environment (Task 2243).

Status: This task is ongoing. The NOAA Marine Debris Program was established in 2005 to investigate and eliminate or dramatically reduce marine debris to protect living marine resources. The Program supports numerous projects that remove debris and derelict fishing gear in areas where leatherbacks are present (see: http://marinedebris.noaa.gov/projects/projects.html#removal). The Sea Turtle Stranding and Salvage Network, formally established in 1980 and the Sea Turtle Disentanglement Network established in 2002 also collect information on entanglement and ingestion of marine debris (see: http://www.sefsc.noaa.gov/species/turtles/strandings.htm).

Recovery Objectives as written in the U.S. Pacific Recovery Plan
The leatherback recovery criteria for delisting are:

1. All regional stocks that use U.S. waters have been identified to source beaches based on reasonable geographic parameters.

    Status: This goal is complete. Stock structure of nesting turtles in the Pacific Ocean has been identified using DNA analysis, flipper tagging, and satellite telemetry. A mixed stock analysis of leatherback turtles along the California coast has been completed. Over 12,500 tissue samples are archived in the NMFS Southwest Fisheries Science Center Molecular Research Sample Collection for use in a variety of population structure, demographic, and trophic ecology studies. For further detail see Section 2.3.1 Biology and Habitat—Taxonomy, Phylogeny, and Genetics.

2. Each stock must average 5,000 (or a biologically reasonable estimate based on the goal of maintaining a stable population in perpetuity) females estimated to nest annually (FENA) over six years.

   Status: Efforts to attain this goal are ongoing. However, key nesting populations (measured by decline in annual nests) are severely declining (see recovery criterium no. 3 below).

3. Nesting populations at "source beaches" are either stable or increasing over a 25-year monitoring period.

   Status: Efforts to attain this goal are ongoing. However, key nesting populations throughout the Pacific and Indian Oceans are declining. Leatherback population trends have been evaluated, and conservation strategies via stochastic simulation models have been designed and evaluated. Monitoring and protection of leatherbacks nesting in Mexico and Costa Rica is ongoing. Currently, all primary nesting beaches in Mexico are protected (although egg poaching still exists), and secondary nesting beaches are partially protected. Playa Grande, Costa Rica, is protected as part of the Las Baulas National Marine Park. Aerial surveys have been conducted to determine abundance and distribution of nesting leatherback turtles in Mexico and Costa Rica. Monitoring and protection of leatherback nesting beaches in the western Pacific, including education of local villagers on the importance of conservation of leatherbacks have been supported. Locations included Papua New Guinea ("no harvest" moratorium set up on Kamiali Beach in 2003 and expanded to seven communities of the Huon Coast in 2005; monitoring index beaches and tagging females; and protecting nest to increase hatchling production), Indonesia (ongoing monitoring and protection, tagging, and telemetry), Solomon Islands (monitoring and nest protection), and Vanuatu (monitoring and nest protection; and surveying for other possible leatherback nesting beaches). For further detail see Section 2.3.1 Biology and Habitat—Abundance and Population Trends.

4. Existing foraging areas are maintained as healthy environments.

   Status: Efforts to attain this goal are ongoing. Stable isotope analyses have been conducted to determine habitat use and foraging strategies. In the western Pacific, several habitat use areas in the South China, Sulu and Sulawesi Seas, Indonesian Sea, Tasman and Coral Seas, East Australian Extension Current, Kuroshio Extension, Equatorial Eastern Pacific, and California Current Ecosystem have been identified. In the eastern Pacific, the South Pacific Gyre is an important migratory and foraging habitat and waters adjacent to Parque Nacional Marino Las Baulas provides important interesting habitat. Over 12,500 tissue samples are archived in the NMFS Southwest Fisheries Science Center Molecular Research Sample Collection for use in a variety of population structure, demographic, and trophic ecology studies. The Stable Isotope Laboratory at NMFS Southwest Fisheries Science Center maintains over 1,500 marine turtle tissue and environmental (i.e. dietary) samples, the vast majority of which have been analyzed for stable-carbon and -nitrogen isotopes.

5. Foraging populations are exhibiting statistically significant increases at several key foraging grounds within each stock region.

   Status: Efforts to attain this goal are ongoing. Aerial surveys have been conducted since the early 1990s to identify foraging areas, distribution and trends of leatherbacks off central and northern California. Aerial and ship-based surveys also were completed to identify foraging hotspots for leatherbacks off Oregon and Washington coasts. The distribution and abundance of leatherback turtles within the coastal California ecosystem has been described. Aerial surveys of the Sulu and Sulawesi seas adjacent to the Philippines have been initiated and are ongoing.

6. A management plan designed to maintain sustained populations of turtles is in place.

   Status: This task is ongoing. A draft NMFS' Western Pacific Leatherback Action Plan has been developed.

7. All priority #1 tasks have been implemented.

   - <u>Eliminate directed take of turtles and their eggs (Tasks 1.1.1.1 and 1.1.1.2)</u>.

   Status: This task is ongoing. NMFS and FWS are party to several international agreements to address directed take and work with in-country partners to support and encourage take reduction. In addition, NMFS supports several outreach efforts to reduce directed take of leatherbacks and their eggs, including those conducted in Mexico, Indonesia, and Vanuatu.

   - <u>Ensure that coastal construction activities avoid disruption of nesting and hatchling activities (Task 1.1.2)</u>.

   Status: This task is ongoing. Monitoring and protection programs for leatherbacks nesting in Mexico, Costa Rica, Papua New Guinea, Indonesia, Solomon Islands, and Vanuatu have been supported.

   - <u>Reduce nest predation by domestic and feral animals (Task 1.1.3)</u>.

   Status: This task is ongoing. Monitoring and protection programs for leatherbacks nesting in Mexico, Costa Rica, Papua New Guinea, Indonesia, Solomon Islands, and Vanuatu have been supported.

   - <u>Collect biological information on nesting turtle populations (Tasks 1.1.5.1, 1.1.5.2, and 1.1.5.3.1 through 1.1.5.3.3)</u>.

   Status: This task is ongoing. A Turtle Research Database System was developed by six international agencies and implemented by the Secretariat of the Pacific Regional Environment Programme (SPREP) to manage tagging data for SPREP member Pacific Island nations. Over 12,500 tissue samples are archived in the NMFS Southwest

Fisheries Science Center Molecular Research Sample Collection for use in a variety of population structure, demographic, and trophic ecology studies. The Stable Isotope Laboratory at NMFS Southwest Fisheries Science Center maintains over 1,500 marine turtle tissue and environmental (i.e. dietary) samples, the vast majority of which have been analyzed for stable-carbon and -nitrogen isotopes.

- Reduce directed take of turtles through public education and information (Task 2.1.1.1) and maintain the enforcement of protective laws on the part of law enforcement and the courts (Task 2.1.1.2).

Status: This task is ongoing. NMFS and FWS are party to several international agreements to address directed take and work with in-country partners to support and encourage harvest reduction and compliance with Regional Fishery Management Organization conservation measures to reduce fishery bycatch. One of many such projects includes implementation of the Marshall Islands Sea Turtle-Fisheries Interaction Outreach Education project to build sea turtle conservation and management capacity of the Marshall Islands Marine Resources Authority (task 2.1.1.1). Outreach programs to educate locals about sea turtle conservation and provide alternatives to directed harvest have been supported in Papua New Guinea, Indonesia, Solomon Islands, and Vanuatu. Research on the costs and benefits of conservation strategies have been supported to better understand the impact these strategies have on local communities. For further detail see Section 2.3.2.4 Inadequacy of existing regulatory mechanisms.

- Determine distribution, abundance, and status in the marine environment and identify threats on foraging grounds (Tasks 2.1.2.1 through 2.1.2.4).

Status: This task is ongoing. Satellite tags were attached to turtles in Papua New Guinea, Indonesia, Solomon Islands, Costa Rica, Colombia, Panama, and United States (California) to gather information regarding migratory movements and pelagic habitat use (task 2.1.2.1).

- Reduce the effects of entanglement and ingestion of marine debris (Tasks 2.1.3.1 through 2.1.3.3).

Status: This task is ongoing. The NOAA Marine Debris Program was established in 2005 to investigate and eliminate or dramatically reduce marine debris to protect living marine resources. The Program supports numerous projects that remove debris and derelict fishing gear in areas where leatherbacks are present (see: http://marinedebris.noaa.gov/projects/projects.html#removal). The Sea Turtle Stranding and Salvage Network, formally established in 1980, also collects information on entanglement and ingestion of marine debris (see: http://www.sefsc.noaa.gov/species/turtles/strandings.htm).

- Monitor and reduce incidental mortality in the commercial and recreational fisheries (Tasks 2.1.4.1 and 2.1.4.2).

Status: The task is ongoing. National observer programs have been supported through in-country fishery observer training to improve sea turtle species identification, reporting, handling, and education regarding fishery mitigation techniques in Palau, Indonesia, Vietnam, Papua New Guinea, Solomon Islands, Vanuatu, Kiribati, Marshall Islands, Federated States of Micronesia, Fiji, Cook Islands, Philippines, and Secretariate of the Pacific Community observers based out of New Caledonia. Other countries have also benefited from NOAA faciliated trainings, such as Tuvalu, Samoa, and Taiwan. An observer program in the Chilean swordfish-directed longline fishery has been supported, and circle hooks and technical support have been provided for experimental testing of modified gear. An observer program in Peru has been supported to document the threat of shark and mahi mahi longline fisheries on leatherback turtles and to document direct harvest. The efficacy of longline gear technology to reduce sea turtle interactions in Pacific Ocean high seas fisheries has been tested in collaboration with Japan. A program in Vietnam was supported to promote sustainable fishing practices in the tuna longline fishery and to strengthen the national observer program as a measure of commitment by Vietnam as a participating non-member to the Western and Central Pacific Fisheries Commission. Five International Fisheries Forums were held to promote the transfer and uptake of commercial longline bycatch reduction technology to international longline fleets of the Pacific. The U.S. Hawaii-based shallow-set swordfish longline fishery has 100% observer coverage, and the deep-set tuna longline fishery has 20-25% observer coverage. Leatherback interaction rates and mortality rates in U.S. Pacific swordfish directed longline fleets have been reduced by requiring specific gear configurations and operational requirements that include use of circle hooks and non-squid bait; fishery closures based on maximum annual turtle interaction limits; area restrictions; proper handling of hooked and entangled turtles; use of disentangling and de-hooking equipment such as dip nets, line cutters, and de-hookers; and reporting sea turtle interactions. Vessel owners and operators are also required to participate in protected species workshops to raise awareness of sea turtle ecology and ensure compliance with sea turtle protective regulations. In addition, since 2001, regulations implementing a large time and area closure off the U.S. west coast have significantly reduced leatherback interactions with the California-based large mesh drift gillnet fishery targeting swordfish/common thresher shark. For further detail see Section 2.3.2.5 Other natural and manmade factors affecting its continued existence.

- Identify and ensure the long-term protection of important marine habitats (Tasks 2.2.1 and 2.2.2).

Status: This task is ongoing. In 2012, critical habitat for leatherbacks was designated off the coast of Washington, Oregon, and California.

- Support existing and develop new international agreements and conventions to ensure that turtles in all life stages are protected in foreign waters (Tasks 4.1, 4.2, and 4.3).

Status: This task is ongoing. NMFS and FWS are party to several international agreements and conventions that are relevant to the conservation and protection of leatherbacks in the Pacific and Indian Oceans, including the Indian Ocean South-East Asian Marine Turtle Memorandum of Understanding, Memorandum of Understanding on Association of South East Asian Nations Sea Turtle Conservation and Protection, Inter-American Convention for the Protection and Conservation of Sea Turtles, and the Secretariat of the Pacific Regional Environment Programme, and Regional Fishery Management Organizations such as the Western and Central Pacific Fisheries Commission and the Inter-American Tropical Tuna Commission.

2.3 Updated Information and Current Species Status

The review is based on information through June 2013. The review does not generate new data through research or modeling. Rather, it provides an overview of the information on leatherback biology, population distribution and trends, habitat, and threats that have emerged since the last 5-year review (NMFS and FWS 2007) to assess whether a status review of the current listing classification for the leatherback sea turtle is appropriate.

Since the 2007 5-year review, we continue to make strides in our knowledge of the biology of leatherbacks, especially away from the nesting beach. Advances in genetic and stable isotope analyses, tagging techniques, such as satellite, radio, and sonic telemetry have vastly improved our knowledge of the biology and ecology of leatherback sea turtles. Important contributions have been made toward hypothesizing the impact of climate and oceanographic processes on the contrasting population trends observed between the Atlantic, Pacific, and Indian Oceans. Increased evaluation of fisheries bycatch worldwide has provided important insights into the management of this species.

2.3.1 Biology and Habitat

Distribution

The leatherback sea turtle is globally distributed. Leatherbacks range as far north as ~ 71° N to 47° S latitude in the southern hemisphere, and they nest from 38° N to 34° S latitude, depending on the ocean basin (reviewed by Eckert *et al.* 2012). In the Atlantic Ocean, they are found as far north as the North Sea, Barents Sea, Newfoundland, and Labrador (Goff and Lien 1988; James *et al*. 2005a; Marquez 1990, Threlfall 1978) and as far south as Argentina and the Cape of Good Hope, South Africa (Hughes *et al*. 1998; Luschi *et al*. 2003b, 2006; Marquez 1990). They also occur in the Mediterranean Sea (Camiñas 1998; reviewed by Casale and Margaritoulis 2010). In the Pacific Ocean, leatherback distribution extends from the waters of British Columbia (McAlpine *et al*. 2004, 2007; Spaven *et al.* 2009) and the Gulf of Alaska (Hodge and Wing 2000) to the waters of Chile and New Zealand (South Island). They also occur throughout the Indian Ocean (Hamann *et al*. 2006a; Nel 2012).

Leatherbacks nest on beaches in the tropics and sub-tropics and forage into higher-latitude sub-polar waters. Important nesting areas in the western Atlantic Ocean occur in Florida, United States; St. Croix, U.S. Virgin Islands; Puerto Rico; Costa Rica; Panama; Colombia; Trinidad and Tobago; Guyana; Suriname; French Guiana; and southern Brazil (Bräutigam and Eckert 2006; Marquez 1990; Spotila *et al*. 1996). Other minor nesting beaches are scattered throughout the Caribbean, Brazil, and Venezuela (Hernández *et al*. 2007; Mast 2005-2006; Velásquez *et al.* 2010). In the eastern Atlantic Ocean, a globally significant nesting population is concentrated in Gabon on the west coast of central Africa (Witt *et al.* 2009). Other widely dispersed but fairly regular nesting occurs between Mauritania in the north and Angola in the south (Fretey *et al*. 2007a). In the Indian Ocean, major nesting beaches occur in South Africa, Sri Lanka, and Andaman and Nicobar islands, with smaller populations in Mozambique, Java, and Malaysia (Hamann *et al*. 2006a; Nel 2012). In the western Pacific Ocean, the main nesting beaches occur in the Solomon Islands, Papua Barat Indonesia, and Papua New Guinea (Dutton *et al.* 2007; Limpus 2002). Lesser nesting occurs in Vanuatu (Petro *et al*. 2007), Fiji (Rupeni *et al*. 2002), and southeastern Australia (Dobbs 2002; Hamann *et al*. 2006a) and is very rare in the North Pacific Ocean (Eckert 1993). In the eastern Pacific Ocean, important nesting beaches occur in Mexico and Costa Rica with scattered nesting along the Central American coast (Marquez 1990). Nesting is very rare in the Gulf of California (Seminoff and Dutton 2007).

Although leatherbacks occur in Mediterranean Sea waters, no nesting is known to take place in this region (Camiñas 1998; reviewed by Casale and Margaritoulis 2010).

Migration

Adult leatherbacks migrate greater distances than adult sea turtles from the family Cheloniidae (Hays and Scott 2013), sometimes travelling up to 11,000 km from their breeding areas (Benson *et al.* 2011). Leatherbacks possess extraordinary navigational skills and are able to travel great distances and return to their breeding and nesting sites after several years away. The actual navigational mechanisms are not known but several factors may underlie a sea turtle's ability to navigate, including magnetic inclination (reviewed by Luschi 2013). Their navigational skills are even more remarkable, given the influence currents may have on their movement through water. For example, leatherbacks off South Africa largely moved with the prevailing currents (Lambardi *et al.* 2008; Luschi *et al*. 2003a); whereas females tracked from Playa Grande, Costa Rica (Shillinger *et al.* 2008) and Guyana, South America (Gaspar *et al*. 2006) were displaced by currents. Given their association with currents during their migration, leatherbacks likely rely on a complex navigation system to travel great distances between breeding and foraging areas (Luschi 2013; Sale and Luschi 2009). One possibility is the turtle makes on-course corrections as it detects current flow (Sale and Luschi 2009). However, Galli *et al.* (2012) found that leatherbacks actively swim in the currents during most of their journey, although with a random orientation with respect to the current, indicating the turtle cannot detect the current. Another possibility is the turtle relies on a large-scale magnetic map to bring them back to the general target area, and then gathers local cues to home in on a nesting beach or foraging area (Mills Flemming *et al.* 2010; Sale and Luschi 2009).

Migration patterns differ by region, driven by local oceanographic process, and multiple migration strategies exist within breeding populations. Migration patterns are described by ocean basin in the following section.

Atlantic Ocean

In the Atlantic Ocean, equatorial waters appear to be a barrier between breeding populations. In the northwestern Atlantic Ocean, post-nesting female migrations appear to be restricted to north of the Equator but the migration routes vary (reviewed by Eckert *et al*. 2012; Saba 2013). For example, Fossette *et al.* (2010a, 2010b) found that turtles tracked from nesting beaches in French Guiana, Suriname, and Grenada and turtles caught in waters off Nova Scotia and Ireland displayed three distinct migration strategies. Leatherbacks made round-trip migrations from where they started through the North Atlantic Ocean heading northwest to fertile foraging areas off the Gulf of Maine, Canada, and Gulf of Mexico; others crossed the ocean to areas off western Europe and Africa; while others spent time between northern and equatorial waters. These data support earlier studies that found adults and subadults captured in waters off Nova Scotia, Canada, stayed in waters north of the Equator (James *et al.* 2005b, 2005c; reviewed by Saba 2013). Females tracked from nesting beaches in Brazil stayed in waters off Brazil, Uruguay, and Argentina (Almeida *et al*. 2011). Adult and subadult leatherbacks caught in fisheries operating in southern waters off Uruguay (Fossette *et al.* 2010a; Lopez-Mendilaharsu *et al.* 2009) and Brazil (Almeida *et al.* 2011) remained in the southwestern Atlantic Ocean. In the eastern Atlantic Ocean, post-nesting females tracked from Gabon exhibit varying dispersal patterns. Satellite telemetry studies show females either remained in highly productive pelagic waters of the equatorial Atlantic (Billes *et al.* 2006b; Fretey *et al*. 2007c; Witt *et al.* 2011); dispersed south along the African continent (Billes *et al*. 2006b; Witt *et al.* 2011); or transited the Atlantic Ocean to forage off coastal areas of southern Brazil, Argentina, and Uruguay (Billes *et al*. 2006a; Witt *et al.* 2011). Post-nesting females from South Africa headed south with the Agulhas current and either stayed in pelagic areas of the South Atlantic Ocean or Indian Ocean (Hughes *et al*. 1998; Luschi *et al*. 2003b, 2006; Robinson *et al.* 2013).

Genetic studies support the satellite telemetry data indicating a strong difference in migration and foraging fidelity between the breeding populations in the northern and southern hemispheres of the Atlantic Ocean (Dutton *et al.* 2013b; Stewart *et al.* 2013). Genetic analysis of rookeries in Gabon and Ghana confirm that leatherbacks from West African rookeries migrate to foraging areas off South America (Dutton *et al.* 2013b). Foraging adults off Nova Scotia, Canada, mainly originate from Trinidad and none are from Brazil, Gabon, Ghana, or South Africa (Stewart *et al.* 2013).

Indian Ocean

Few data exist on the foraging grounds and migratory corridors of leatherbacks in the Indian Ocean and Southeast Asia region, although leatherbacks have been reported from the waters of 32 of the 44 countries comprising this region (Hamann *et al*. 2006a). As discussed above, leatherbacks nesting in South Africa sometimes travel around the Cape of Good Hope into southeast Atlantic waters (Hughes *et al.* 1998; Luschi *et al*. 2003b, 2006; Robinson *et al.* 2013). Several post-nesting females tracked from iSimangaliso Wetland Park, South Africa, stayed in the Indian Ocean coastal waters off Africa foraging and rarely moving beyond 100 km from shore (Robinson *et al.* 2013). Leatherbacks off South Africa moved largely with the prevailing currents (Lambardi *et al.* 2008; Luschi *et al*. 2003a).

Pacific Ocean
In the western Pacific Ocean, leatherbacks nest year round and migration strategies vary (reviewed by Saba 2013). Benson *et al.* (2011) satellite tagged 126 leatherbacks from the western Pacific population. Tagged turtles were from nesting beaches (Jamursba-Medi and Wermon in Indonesia, Huon Gulf of Papua New Guinea, and Solomon Islands) (n = 44 summer and 45 winter boreal nesters) and foraging grounds off California (n = 27 female and 10 male adults). For boreal summer nesters, the most common migration was into the North Pacific Ocean either to the Kuroshio Extension region or the California Current Ecosystem. The second most common migration was west into the Sulu, Sulawesi, and South China Seas, adjacent to Malaysian Borneo and Palawan Islands, Philippines. Only one female traveled north into the Sea of Japan. The boreal winter nesters went south through the Coral Sea into the high-latitude of the South Pacific Ocean or Tasman Sea. However, one female moved west through the Coral Sea into the Gulf of Papua. Among foragers tagged in coastal waters off California, the majority moved north and spent time in areas off northern California and Oregon, before moving towards the equatorial eastern Pacific, then eventually westward presumably towards western Pacific Ocean nesting beaches (Benson *et al.* 2011). The greatest distance travelled was over 11,000 kilometers and took up to a year to complete (Benson 2011-2012; Benson *et al*. 2007b, 2011). Finally, the western Pacific breeding population also migrates to foraging areas in the southeastern Pacific. Genetic analyses of juvenile and adult leatherbacks caught in fisheries off Peru and Chile show a proportion originate from the western Pacific Ocean rookeries (Donoso *et al*. 2000; Dutton 2005-2006, 2006; Dutton *et al.* 2010; Dutton *et al.* 2013a).

In the eastern Pacific Ocean, studies show that females primarily migrate southward to the southern hemisphere into the South Pacific Gyre in pelagic waters off Peru and Chile (Donoso *et al*. 2000; Dutton 2005-2006; Shillinger *et al.* 2008, 2010, 2011). Bycatch data in Peruvian coastal artisanal fisheries indicate leatherbacks are present in coastal areas (Alfaro-Shigueto *et al.* 2007, 2011). Genetic work has also shown that leatherbacks from the eastern Pacific populations, as well as western Pacific populations, are recorded in the North Pacific Ocean (Dutton *et al*. 1998, 2000b, 2002, 2006; Dutton 2005-2006).

Internesting Movement
During the nesting season, females generally stay within 100 km of the nesting beach but also undergo long distances between nesting events, traveling up to 4,500 km during the entire nesting season, (reviewed by Eckert *et al*. 2012). Internesting movements have been described from several nesting beaches (Almeida *et al.* 2011; Benson *et al*. 2007a, 2011; Billes *et al*. 2006b; Eckert 2006; Eckert *et al*. 1996; Eguchi *et al*. 2006b; Fosette *et al*. 2006, 2009; Fulton *et al*. 2006; Hitipeuw *et al*. 2007; Meylan *et al.* 2013; Myers and Hays 2006; Reina *et al*. 2005; Shillinger *et al*. 2006, 2010; Wallace *et al*. 2005; Witt *et al*. 2008). For example, females from nesting beaches in Brazil dispersed up to 160 km from the nesting beach using an area of 4,400 km^2. Foraging areas were identified in waters off Brazil, Uruguay, and Argentina (Almeida *et al.* 2011). In the western Pacific Ocean population, leatherbacks generally stayed within 300 km or less from nesting beaches in Indonesia (Jamursba-Medi, Wermon, Papua Barat), Papua New Guinea, and the Solomon Islands (Benson *et al.* 2011).

Hatchling Dispersal

Little is known about the early life history of leatherbacks from hatchling to adulthood. However, new technologies have been developed to elucidate hatchling dispersal. Passive drifter models have been used to predict the trajectories of hatchlings offshore (e.g., Gaspar *et al.* 2012; Hamann *et al.* 2011; Shillinger *et al.* 2012). Passive drifter model predictions, combined with analysis of sighting, genetic, bycatch and satellite tracking information, indicate hatchlings emerge from nesting beaches in Jamursba-Medi, Indonesia, and Kamiali, Papua New Guinea, and are entrained by highly variable oceanic currents into the North Pacific, South Pacific, or Indian Oceans (Gaspar *et al.* 2012). After 1 to 2 years, these currents may take small juveniles into temperate regions where water temperatures in winter drop well below the minimum temperature likely tolerated by such small individuals. Eckert (2002) summarized the records of nearly 100 sightings of juvenile leatherbacks and found that animals less than 100 cm curved carapace length (CCL) are generally found in water warmer than 26˚C indicating that the first part of a leatherback's life is spent in tropical waters. Gaspar *et al.* (2012) hypothesize that after an initial period of mostly passive drift, juveniles begin to actively swim towards warmer latitudes before winter and back again towards higher latitudes during spring. This simulated migration pattern is used by adult leatherbacks from Jamursba-Medi and Kamiali (Gaspar *et al.* 2012). Scientists have theorized that an adult's choice of migration patterns are influenced by the currents they experienced as a hatchling—known as the "hatchling drift scenario" (reviewed by Saba 2013).

Other technologies are being developed and tested to track leatherback hatchlings. Gearheart *et al.* (2011) tracked hatchlings departing beaches of Papua's Bird's Head Peninsula, Indonesia, using both acoustic and VHF radio tags and found the acoustic tags performed better than the VHF tags, which had poor directionality. Thums *et al.* (2013) used active and passive acoustic monitoring of flatback turtle hatchlings, which they felt showed great potential as a means to understand the in-water behaviour of turtle hatchlings.

Understanding where hatchlings disperse and grow and how it influences their adult migration strategies (e.g., why do females from the western Pacific Ocean make transoceanic journeys to feed in highly productive areas off California while more approximate nesting females from Costa Rica ignore it?) is an essential component to recovering the species.

Demography

Survival

Reliable estimates of survival or mortality at different life history stages are not easily obtained. The annual survival rate for leatherbacks that nested at Playa Grande, Costa Rica, was estimated to be 0.654 for 1993-1994 and 0.65 for those that nested in 1994-1995 (Spotila *et al*. 2000). Rivalan *et al*. (2005) estimated the mean annual survival rate of adult leatherbacks in French Guiana to be 0.91. Pilcher and Chaloupka (2013) used capture-mark-recapture data for 178 nesting leatherbacks tagged at Lababia beach, Kamiali, on the Huon Coast of Papua New Guinea over a 10-year austral summer nesting period (2000-2009). Annual survival probability (ca. 0.85) was constant over the 10-year period. Annual survival was lower than those estimated for Atlantic rookeries (Dutton *et al*. 2005; Rivalan *et al*. 2005). However, the reason for the lower annual survival rate is unknown and may be due to several factors such as greater anthropogenic impacts or lower site fidelity (Pilcher and Chaloupka 2013). For the St. Croix, U.S. Virgin Islands population, the annual survival rate was approximately 0.893 (confidence interval = 0.87-0.92) for adult female leatherbacks at St. Croix (Dutton *et al*. 2005). Annual juvenile survival rate for St. Croix was estimated to be approximately 0.63, and the total survival rate from hatchling to first year of reproduction for a female hatchling was estimated to be between 0.004 and 0.02, given assumed age at first reproduction between 9 and 13 (Eguchi *et al*. 2006). In Florida, annual survival for nesting females was estimated to be 0.956 (Stewart 2007). Spotila *et al*. (1996) estimated the first year (from hatching) of survival for the global population to be 0.0625.

Growth and Age at Maturity

Leatherbacks grow rapidly (approximately 32 cm in carapace length each year) from hatchling to juvenile size, which is relatively faster than other sea turtle species and surprising given leatherbacks subsist on low caloric prey (Jones *et al.* 2011). Extremely rapid growth may be possible because leatherbacks have evolved a mechanism that allows fast penetration of vascular canals into the fast growing cartilaginous matrix of their bones (Rhodin *et al.* (1996). However, it has not been determined if the vascularized cartilage in leatherbacks serves to facilitate rapid growth or affect some other physiological function.

Age at sexual maturity based on skeletochronological data suggest that leatherbacks in the western North Atlantic Ocean may not reach maturity until 29 years of age (Avens and Goshe 2008; Avens *et al.* 2009). The skeletochronological data contradict other estimates (Dutton *et al*. 2005: 12-14 years; Jones *et al.* 2011: 7-16 years; Pritchard and Trebbau 1984: 2-3 years; Rhodin 1985: 3-6 years; Zug and Parham 1996: average maturity at 13-14 years for females). Age at maturity remains a very important parameter to be confirmed as it has significant implications for management and recovery of leatherback populations.

Reproductive Capacity

Clutch frequency per year ranges between 5 and 7 with a maximum observed frequency of 13 (reviewed by Eckert *et al.* 2012). The average number of eggs per clutch varies by region: Atlantic Ocean (85 eggs), western Pacific Ocean (85 eggs), eastern Pacific Ocean (65 eggs) and Indian Ocean (>100 eggs) (reviewed by Eckert *et al.* 2012). The remigration interval averages between 2 and 3 years, but can be longer likely due to environmental conditions (reviewed by Eckert *et al.* 2012). Breeding has been documented to span an average 16 (up to 19) years in South Africa (Nel *et al.* 2013) and 19 years in the U.S. Virgin Islands (reviewed by Eckert *et al.* 2012).

Despite high fecundity, hatching success is lower than other sea turtle species and is attributed to many factors including compromised nesting beach habitat (e.g., erosion, temperature extremes, armament) environment, and handling of the eggs (reviewed by Eckert *et al.* 2012). For example, in Indonesia, low hatching success is due to (1) predation of eggs and hatchlings by introduced pigs and dogs (Bhaskar 1987 in Tapilatu *et al.* 2013; Hitipeuw and Maturbongs 2002; Maturbongs 2000; Suganuma 2006; Suganuma *et al.* 2005), (2) beach erosion (Bhaskar 1987 in Tapilatu *et al.* 2013; Hitipeuw *et al.* 2007), and (3) elevated sand temperatures (Tapilatu and Tiwari 2007). Reproductive experience also may be a factor in hatching success. For example, Rafferty *et al.* (2011) found that remigrant females (i.e., a female who has been recorded to nest at a particular beach and has returned in a subsequent year to nest) at Playa Grande, Costa Rica, arrived earlier, produced more clutches, and had higher hatching success than neophytes. Also, some individual females consistently laid nests with higher hatch success. Plot *et al.* (2012) found that highly productive females had longer telomeres (i.e., repetitive non-coding DNA sequences that cap the ends of chromosomes and shorten through successive cell division until they reach a critical length causing instability, cell senescence and ultimately cell death). Telomere length may be an easily accessible marker of individual reproductive quality in female leatherback turtles (Plot *et al.* 2012).

Sex Ratios
A comparison of hatchling sex ratios at several nesting beaches in the Atlantic and Pacific Oceans suggests that the Pacific populations may be more female biased (Binckley *et al*. 1998) than Atlantic populations (Godfrey *et al*. 1996; Turtle Expert Working Group 2007). However, caution is warranted about making basin wide comparisons. Few studies have been conducted in the Pacific and sex ratios varied by beach and even clutch (Binckley *et al*. 1998), or the sample size was small (Steckenreuter *et al.* 2010). Other studies support a more narrow temperature regime for sex determination in the Atlantic Ocean. Chevalier *et al*. (1999) compared temperature-dependent sex determination patterns between the Atlantic (French Guiana) and the Pacific (Playa Grande, Costa Rica) and found that the range of temperatures producing both sexes was significantly narrower for the Atlantic population.

Sex ratios in hatchlings may not accurately reflect the sex ratios in later life stages due to the possibility of differential mortality. Stewart and Dutton (2011, 2011-2012) inferred paternity from genetic samples of hatchlings from known females at Sandy Point National Wildlife Refuge, U.S. Virgin Islands. They found that 46 females mated with 47 individual males suggesting the operational sex ratio is more balanced in later life. However, an analysis of strandings along the U.S. Atlantic and Gulf of Mexico coast from 1980-2004 indicates a female bias (60%) in subadults and adults (Turtle Expert Working Group 2007). The proportion of females overall appears to have increased in the strandings since the 1980s, but this pattern is less evident when evaluated by region (i.e., north and south Atlantic and Gulf). In Canada, Atlantic Ocean, the sex ratio was 69% female for turtles greater than 145 cm CCL (James *et al.* 2007). Brazil also had a female biased sex ratio (Barata *et al.* 2004); whereas, in the Mediterranean, United Kingdom waters, and along Atlantic France, overall there was no strong female bias among strandings, sightings, and captures (Turtle Expert Working Group 2007). A balanced sex ratio in the adult population has been reported for other species of sea turtle (Hays *et al*. 2010).

Taxonomy, Phylogeny, and Genetics
The leatherback taxonomic classification (below) is unchanged since the last 5-year review (NMFS and FWS 2007).

Kingdom:	Animalia
Phylum:	Chordata
Class:	Reptilia
Order:	Testudines
Family:	Dermochelyidae
Genus:	*Dermochelys*

Species: *coriacea*
Common name: Leatherback sea turtle

The leatherback is unique among sea turtles because it is the only extant survivor of an evolutionary lineage that diverged from other sea turtles 100-150 million years ago (Zangerl 1980). Extinctions during the Pleistocene glaciations most likely reduced leatherbacks to a single lineage (Dutton 2004; Dutton *et al.* 1999). Although leatherbacks have a deeper evolutionary origin than the other extant sea turtle species, analysis of genetic data suggest a relatively recent global radiation (Bowen and Karl 1996; Dutton *et al*. 1996, 1999). Analysis of maternally inherited mitochondrial DNA (mtDNA) indicates an ancestral separation between the Atlantic and Indo-Pacific Ocean populations of 0.17 million years before present (Duchene *et al.* 2012). The post-Pleistocene recolonization of the Atlantic Ocean most likely occurred via the eastern Atlantic as nesting populations in Ghana and Gabon share haplotypes with populations in the Indo-Pacific (Dutton *et al.* 2013b).

Leatherbacks exhibit low genetic diversity in the mitochondrial genome (Dutton *et al*. 1996, 1999; see Jensen *et al.* 2013). The most divergent mtDNA haplotypes occur between the western Atlantic Ocean (Florida, Costa Rica, Trinidad, French Guiana/Suriname, St. Croix) and the eastern Pacific Ocean (Costa Rica, Mexico) (Dutton *et al*. 1999). Hypotheses for low genetic diversity include population bottlenecks due to recent extinction, selection pressure that led to the replacement of recent ancestral mtDNA, and insufficient time to accumulate new mutations at the population level (Dutton *et al*. 1999). Furthermore, low genetic diversity may be linked to infrequent or no multiple paternity within or among successive clutches of a female (Crim *et al*. 2002; Curtis 1998; Dutton and Davis 1998; Dutton *et al*. 2000a; Rieder *et al*. 1998) suggesting that perhaps females rarely encounter multiple males or that sperm competition may occur (Dutton *et al*. 2000a). However, females nesting in Gandoca-Manzanillo Wildlife Refuge, Costa Rica, mated with multiple partners, and there was evidence of mating with new mates between nesting events (Figgener *et al.* 2012). Stewart and Dutton (2011, 2011-2012) found five of 12 females nesting in St. Croix, U.S. Virgin Islands, had mated with more than one male. However, unlike the Costa Rica study, they found the individual female's breeding partners contributed to all clutches throughout the nesting season, indicating that she mated prior to (not during) the nesting season and stored the sperm (Stewart and Dutton 2011). A previous study (Dutton *et al*. 2000a) at the site showed no evidence of multiple paternity, which may have been missed due to a smaller sample size (Stewart and Dutton 2011). Multiple paternity may be linked to population abundance (i.e., the nesting population at St. Croix is increasing at about 13% per year), which would increase the likelihood of encountering and mating with multiple partners, but additional studies are needed at other dense nesting sites to validate this theory (Stewart and Dutton 2011).

In the Atlantic Ocean, Dutton *et al.* (2013b) found a higher degree of fine-scale population differentiation than had been detected with the less informative mtDNA marker in previous studies (Dutton 1995; Dutton *et al.* 1999). Dutton *et al.* (2013b) conducted a comprehensive genetic re-analysis of rookery stock structure using longer (more informative) mtDNA sequences combined with nuclear marker data from 17 microsatellite loci with larger sample sizes and previously unsampled rookeries in the Atlantic and southwest Indian Ocean. Nesting sites included Brazil, Costa Rica, French Guiana/Suriname, Gabon, Ghana, South Africa, Trinidad, United States (Florida), and U.S. Virgin Islands (St. Croix). They found sufficient genetic differentiation with the nuclear markers to suggest all nine of these rookeries represent demographically independent populations (DIPs) or Management Units (MUs) (representing fine scale population structure shaped by environmental or behavioral processes on ecological rather than evolutionary timescales; see Dutton *et al.* 2013b). Significant mtDNA differentiation was found for all populations except between Florida and Costa Rica and between Trinidad and French Guiana/Suriname, indicating recent shared ancestry for these groups. Dutton *et al.* (2013b) suggested that the mtDNA homogeneity between Florida and Costa Rica indicate Costa Rica may be a source population for the growing Florida population. They also concluded that the genetic differentiation with nuclear markers found among rookeries that were homogenous with regard to mtDNA suggests that breeding site fidelity by males may also contribute to delineation of rookeries, and that male-mediated gene flow may not be as pronounced as previously thought (Dutton *et al.* 2013b; see Jensen *et al.* 2013). Despite these two exceptions, the prevalence of significant mtDNA differentiation between rookeries throughout the Atlantic Ocean indicate that natal homing in leatherbacks may be more precise than previously reported (Dutton *et al*. 1999, 2007). In addition to the degree of site fidelity exhibited in males and females, other factors such as colonization events and biased sex ratios may influence population substructuring (Dutton *et al.* 2013b).

Dutton *et al.* (2013b) results support earlier genetic, satellite telemetry, and tagging studies indicating demographic separation in some of the Atlantic Ocean rookeries (Billes *et al.* 2006a; Dutton *et al*. 2003; LaCasella and Dutton 2008; Turtle Expert Working Group 2007; Vargas *et al*. 2008; Witt *et al.* 2011) and more recent studies (Carreras *et al.* 2013; Molfetti *et al*. 2013; Richardson *et al.* 2013; Wallace *et al.* 2010b). However, further sampling at nesting sites is needed throughout the Caribbean and West Africa to understand finer scale population structuring (Dutton *et al.* 2013b). For example, genetic analysis of leatherbacks nesting in the Dominican Republic show a significant differentiation from nesting populations in St. Croix, French Guiana and Trinidad (Carreras *et al.* 2013). Further, resightings of flipper-tagged nesting females between Panama, Columbia, Venezuela, and Guyana blur the population boundary between the two distinct rookeries in Costa Rica and French Guiana/Suriname, which are at the extreme edges of the regional stock (Dutton *et al.* 2013b). Some females from Honduran and Colombian beaches were discovered on beaches in Costa Rica (Troëng *et al*. 2004) suggesting one large rookery along the entire coastline. Four leatherbacks tagged on the beaches of Costa Rica and Panama were later found nesting in Cuba, Florida, St. Croix, and Grenada, thereby weakening the concept of a distinct Western Caribbean leatherback population. A female tagged on St. Croix nested in Dominica, and a leatherback turtle tagged in Costa Rica was later found on a beach in the Indian River Lagoon, Florida (reviewed by Bräutigam and Eckert 2006; Turtle Expert Working Group 2007). Important rookeries in West Africa including Bioko Island in Equatorial Guinea, smaller nesting populations in Ivory Coast, northern Gabon, Congo, and Angola have not been sampled (Dutton *et al.* 2013b).

In the Pacific Ocean, genetic studies have identified three distinct populations (referred to also as genetic stocks or Management Units; see Wallace *et al.* 2010b) of leatherback turtles: (1) Mexico and Costa Rica, which are genetically homogenous but distinct from the western populations; (2) Papua Barat in Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu, which comprise a metapopulation representing a single genetic stock; and (3) Malaysia (Barragan *et al*. 1998; Barragan and Dutton 2000; Dutton 2005-2006, 2006; Dutton *et al.* 1999, 2000b, 2007). The genetically distinct Malaysia nesting population likely is extirpated (Chan and Liew 1996; Dutton 2005-2006; Dutton *et al*. 1999).

In the Indian Ocean, a significant gap in knowledge remains concerning the genetic population structure of leatherback rookeries. Published genotypes only exist for Malaysia, Indonesia, and South Africa (Dutton *et al*. 1999, 2007). It has been hypothesized that the nesting beaches in Sri Lanka and the Nicobar Islands might be part of a distinct Indian Ocean population (Dutton 2005-2006). Genetic samples were taken from females nesting at Little Andaman Island, India, from 2008 through 2010, but results have not been published (Namboothri *et al.* 2010). Further genetic sampling has been recommended for all the Andaman and Nicobar islands, as well as northern and eastern Australia, Mozambique, Sri Lanka, Sumatra, Java, Thailand, and Vietnam (Dutton *et al*. 1999, 2007).

In the Mediterranean Sea, nesting has not been documented (Camiñas 1998; reviewed by Casale and Margaritoulis 2010). Leatherbacks in Mediterranean Sea waters originate from the Atlantic Ocean populations (P. Dutton, NMFS, unpublished data).

Habitat Use or Ecosystem Conditions

Marine

As described earlier, leatherbacks inhabit waters as far north as ~ 71° N and as far south as 47° S latitude. Leatherbacks have evolved physiological, anatomical, and behavioral adaptations (Bostrom *et al.* 2010; Casey *et al.* 2012; Fossette *et al.* 2009; Frair *et al*. 1972; Greer *et al*. 1973; reviewed by Southwood Williard 2013) that allow them to exploit waters far colder than any other sea turtle species would be capable of surviving. Thus, leatherbacks are able to take advantage of a wide variety of marine ecosystems (reviewed by Saba 2013; see NOAA large marine ecosystem website: http://www.lme.noaa.gov/). Within these ecosystems, various oceanic features such as water temperature, downwelling, Ekman upwelling, seasurface height, chlorophyll-a concentration, and mesoscale eddies affect the presence of leatherbacks (Bailey *et al.* 2013; Benson *et al.* 2011). The physical characteristics observed within these marine ecosystems also affect the distribution and abundance of leatherback prey (reviewed by Saba 2013). Leatherbacks mainly eat gelatinous organisms, particularly of the class Scyphozoa, but other taxa including crustaceans, vertebrates, and plants are ingested (reviewed by Eckert *et al.* 2012; Dodge *et al.* 2011; Jones and Seminoff 2013). Because leatherbacks must consume large amounts of food to meet their energetic demands (Heaslip *et al.* 2012; Jones *et al.* 2012), it is important that they have access to areas of high productivity.

Satellite telemetry and stable isotope studies underscore the importance of the association of leatherback presence in highly productive ecosystems. For example, in the Bay of Biscay along the French Atlantic coast, vertical mixing of the continental slope waters and the topographic effects of the banks within the Bay concentrate plankton and other prey species providing optimal conditions for foraging habitat (reviewed by Zaldua-Mendizabal *et al.* 2013). A satellite-tagged leatherback caught in fisheries off the southwest coast of Ireland moved south and spent 66 days apparently foraging in the rich waters west of the Bay of Biscay (Doyle *et al.* 2008). In the low nutrient areas of the North Atlantic Gyre and the Sargasso Sea, leatherbacks transited the area at high speed until they reached more productive areas at high latitudes where they foraged (Fossette *et al.* 2010b).

For the western Pacific population, seven ecoregions (South China/Sulu and Sulawesi Seas, Indonesian Seas, East Australian Current Extension, Tasman Front, Kuroshio Extension, equatorial Eastern Pacific, and California Current Extension) were identified as important seasonal foraging areas (Benson *et al.* 2011). Off the U.S. west coast, two areas were identified as essential ("critical") habitat for leatherbacks in 2012. One includes the nearshore waters between Cape Flattery, Washington, and Cape Blanco, Oregon extending offshore to the 2000 meter isobaths. This area was identified as the principal Oregon/Washington foraging area and included important habitat associated with the Columbia River Plume, and Heceta Bank, Oregon. Here, great densities of primary prey species, brown sea nettle, occur seasonally north of Cape Blanco (Reese 2005; Suchman and Brodeur 2005; Shenker 1984). The second area identified as "critical habitat" includes offshore waters between the 200 and 3000 meter isobaths from Point Arena to Point Sur, California and waters between the coastline and the 3000 meter isobath from Point Sur to Point Arguello, California. Here, the neritic waters between Point Sur and Point Arguello are strongly influenced by coastal upwelling processes that produce abundant and dense aggregations of leatherback prey. The southern portion of the region includes Morro and Avila Bays where large densities of brown sea nettles have been observed seasonally in fisheries monitoring surveys and trawl surveys. Telemetry data analyzed by Benson *et al.* (2011) indicate that leatherbacks forage in this area.

In the eastern Pacific Ocean, post-nesting females from Playa Grande, Costa Rica, commonly forage offshore in the South Pacific Gyre in upwelling areas of cooler, deeper water and high productivity (Shillinger *et al.* 2011). During the nesting season, they stay within the shallow, highly productive, continental shelf waters (Shillinger *et al.* 2010).

Stable isotope analysis can complement satellite data of leatherback movements and identify important foraging areas (reviewed by Jones and Seminoff 2013; Seminoff *et al.* 2012). Leatherback gelatinous prey show isotopic values that reflect regional food webs, and leatherbacks retain these values in their soft tissue long after they depart the foraging area (Seminoff *et al.* 2012). For example, nitrogen isotope values in skin samples (n = 65) collected from females returning to nest on Jamursba-Medi, Papua, Indonesia, showed distinct low and high nitrogen values. Known satellite-tracked females (n= 13; Benson *et al.* 2011) arriving from the western Pacific and eastern Pacific foraging sites had similar disparate nitrogen values (Seminoff *et al.* 2012). The differences in nitrogen values were due to baseline values of the primary producers between the two eastern and western Pacific Ocean broad foraging areas and not a difference in diet (Seminoff *et al.* 2012). The western Pacific foraging areas are dominated by source nitrogen with a lower isotopic composition; whereas, eastern Pacific foraging areas are generally characterized by higher baseline stable nitrogen values in surface waters, owing to dentrification in the eastern Tropical Pacific and northward flow of the current to leatherback foraging grounds off the U.S. West Coast (Seminoff *et al.* 2012). Similar isotopic patterns were found between foraging grounds in the northern and eastern tropical areas in the Atlantic Ocean (Caut *et al.* 2008; Seminoff *et.al.* 2013). This relatively high stable nitrogen isotope pattern was also found by Wallace *et al.* (2006b) in their comparison of leatherback turtles nesting in Pacific Costa Rica versus St. Croix, U.S. Virgin Islands. In the western North Atlantic, Dodge *et al.* (2011) used a stable isotope analysis that revealed levels of carbon and nitrogen in small juvenile leatherbacks that are characteristic of offshore food webs, suggesting offshore foraging areas (e.g., *Sargassum*) are important to the early life stage of the leatherback.

Terrestrial

Nesting beach habitat is generally associated with deep water and strong waves and oceanic currents, but shallow water with mud banks are also used by leatherbacks (Turtle Expert Working Group 2007). Beaches with coarse-grained sand and free of rocks, coral or other abrasive substrates appear to be selected (reviewed by Eckert *et al.* 2012). Leatherbacks may have an impact on the nutrient loads on beaches where nest density is high (Davenport 2011; Pollock *et al.* 2012). Davenport (2011) estimated that 224–782 metric tons of leatherback eggs are laid each year in Gabon, equating to 0.37–1.3 metric tons per kilometer of coastline. At this density, the eggs will provide nutrition for an extensive terrestrial food web, either directly or indirectly.

Abundance and Population Trends

Historical descriptions of leatherbacks are rarely found in the accounts of early sailors, and the size of their population before the mid-20th century is speculative. Even for large nesting assemblages like French Guiana and Suriname, nesting records prior to the 1950s are lacking (Rivalan *et al*. 2006). By the 1960s, several nesting sites were being discovered in the western Atlantic, Pacific Mexico, and Malaysia. Soon after, other populations in Pacific Costa Rica and Mexico were identified. The lack of historical published nesting accounts for these large reptiles may be due to a lack of publicity by indigenous people or lack of human habitation along leatherback nesting beaches. Today, nesting beaches are known in all major ocean basins with catastrophic declines observed in the eastern Pacific (Spotila *et al*. 2000), Malaysia (Chan and Liew 1996), and Indonesia (Tapilatu *et al.* 2013).

Pritchard (1982) estimated 115,000 females occurred worldwide, of which 60% nested along the Pacific coast of Mexico. Spotila *et al*. (1996) later estimated that only 34,500 females (with confidence limits of 26,200 to 42,900 females) remained worldwide. However, the most recent population size estimate for the North Atlantic alone is a range of 34,000-94,000 adult leatherbacks (Turtle Expert Working Group 2007). Abundance and population trends (specified by either nesting population or total population where known) are summarized by each ocean basin below. Table 1 provides site location information for beaches described in the text, where multiple-year surveys were conducted or trends are known.

Atlantic Ocean

Trends and abundances are provided below for leatherback populations or groups of populations in the Atlantic Ocean. Excluding Africa, 470 nesting sites have been identified of which 58% are small rookeries with less than 25 crawls each year (Dow Piniak and Eckert 2011). Although some authors have independently presented their analyses of trends, and we have included them in the sections below, the Turtle Expert Working Group (2007) undertook trend analyses (regression and Bayesian) on Atlantic populations with a minimum of 10 years of nesting data and those results are included as well. Overall, an increasing or stable population trend is seen in all regions except the Western Caribbean and West Africa (for the latter, no long-term data are available) (Turtle Expert Working Group 2007).

In Florida, United States, the number of nests has been increasing by 10.2% (range 3.1%-16.3%) annually since 1979 (Stewart *et al.* 2011a). The estimate is based on nest counts from 68 beaches from 1979 through 2008 conducted by the Florida Statewide Nesting Beach Survey program. The average annual number of nests in the 1980s was 63 nests, which rose to 263 nests in the 1990s and to 754 nests in the 2000s (Stewart *et al.* 2011a). In 2012, 515 leatherback nests were recorded on the index beaches and 1,712 nests were recorded statewide (http://myfwc.com/research/wildlife/sea-turtles/nesting/). Included in the statewide survey are the Archie Carr and Hobe Sound National Wildlife Refuges. In the 1980s, leatherbacks rarely nested in the Archie Carr National Wildlife Refuge, but by the mid-1990s nesting began to increase with 11 to 52 nests reported annually (Bagley *et al.* 2013). Nest numbers at Hobe Sound National Wildlife Refuge have fluctuated from 2005-2013 with a low of 35 in 2006 and a high of 128 in 2010 (B. Miller, FWS, unpublished data).

In Puerto Rico, the main nesting areas are at Fajardo on the main island of Puerto Rico and on the island of Culebra. Between 1978 and 2005, nesting increased in Puerto Rico from a minimum of 9 nests recorded in 1978 and to a minimum of 469-882 nests recorded each year between 2000 and 2005 (R. Martinez, Department of Natural and Environmental Resources of Puerto Rico, unpublished data). Theannual population growth rate was estimated to be 1.10 with a growth rate confidence interval between 1.04 and 1.12 using nest numbers between 1978 and 2005 (Turtle Expert Working Group 2007). However since 2004, nesting has steadily declined in Culebra (Diez *et al.* 2010; Ramírez-Gallego *et al.* 2013). In 2012, only 5 females nested on the island, which is the lowest recorded since 1993 (C. Diez, Department of Natural and Environmental Resources of Puerto Rico, unpublished data). However, evidence exists that females may be selecting other beaches (Ramírez-Gallego *et al.* 2013). Overall increases are recorded for mainland Puerto Rico and St. Croix, U.S. Virgin Islands, which may indicate that the decline in Culebra is not a true loss to the breeding population but rather a shift in nesting site (Diez *et al.* 2010; Ramírez-Gallego *et al.* 2013).

In the U.S. Virgin Islands, Sandy Point National Wildlife Refuge on the island of St. Croix has been monitored since 1977. The Sandy Point National Wildlife Refuge has the most complete and consistent leatherback nesting data set in the Caribbean. Dutton *et al*. (2005) estimated a population growth of approximately 13% per year on Sandy Point National Wildlife Refuge from 1994 through 2001. Between 1990 and 2005, the number of nests recorded has ranged from a low of 143 in 1990 to a high of 1,008 in 2001 (Garner *et al.* 2005). The average annual growth rate was calculated as approximately 1.10 (with an estimated confidence interval between 1.07 and 1.13) using the number of observed females at Sandy Point, St. Croix, from 1986 to 2004 (Turtle Expert Working Group 2007). However, trends since 2001 suggest the population may be declining, possibly due to a decrease in the number of new nesters, lowered productivity (number of clutches per season and lower hatch success), and an increase in remigration intervals (Garner 2012; Garner and Garner 2010; Garner *et al.* 2012).

In the British Virgin Islands, annual nest numbers from 1986 to 2006 have increased from 0-6 nests per year in the late 1980s to 35-65 nests per year in the 2000s (McGowan *et al.* 2008). Annual growth rate was estimated to be approximately 1.2 for nests laid between 1994 and 2004 (Hastings 2003; Turtle Expert Working Group 2007). The increase in leatherback nests in the British Virgin Islands is likely due to a moratorium on the harvest of females and eggs implemented in 1986, but may also represent individuals from nesting sites throughout the Caribbean recruiting to the British Virgin Islands (McGowan *et al.* 2008).

There are many locations in the Caribbean that cannot be assigned to a particular population due to lack of nesting surveys and genetic sampling. In the insular Caribbean, 0-25 nests are estimated per year in Antigua, Bahamas, Barbados, Bonaire, Cayman Islands (Grand Cayman, Cayman Brac, and Little Cayman), Cuba, Curaçao, Jamaica, Monserrat, Saba, St. Barthelemy, St. Maarten, St. Martin, and Turks and Caicos. Between 25 and 100 nests are estimated annually in Anguilla, Aruba, Dominica, Guadeloupe, and St. Eustatius. Between 100 and 500 nests are estimated per year in Grenada, St. Kitts and Nevis, St. Lucia, and St. Vincent and the Grenadines (Eckert and Bjorkland 2004). Levera Beach, a major nesting beach in Grenada, generally receives 200-900 nesting activities per year, and in 2005, 237 nests were recorded (Maison *et al.* 2010). In Martinique, 150-200 nests are estimated each year (Turtle Expert Working Group 2007). In the Dominican Republic, Jaraqua National Park, the number of nests between 2006 and 2009 averaged 127 (± 88) per year, and surveys outside the Park indicated about 25 nests were laid each year (Tomás *et al.* 2013). No trend data are available because the time series are too short.

For Nicaragua, Lagueux and Campbell (Wildlife Conservation Society, personal communication 2013) provide the following: "Leatherback nesting occurs along the southeast coast of Nicaragua, from the Nicaragua/Costa Rica border northward to the Karaslaya river mouth, a distance of approximately 42 km, although the majority of nesting occurs within 15 km of the border (to the Cangrejera settlement). In 2000, and from 2008 to 2013, an average 80 ± 28.6 clutches were counted (range = 42 clutches (2013) to 132 clutches (2009), however, nesting levels are most likely underrepresented because the border section, which accounts for about 22% of nesting activity, was not surveyed from 2011 to 2013 because of heightened military presence in the area due to a border dispute with Costa Rica (Lagueux and Campbell 2005; Lagueux & Campbell unpublished data; Lagueux *et al.* 2012). Additionally, the Karaslaya section was not included in monitoring surveys until the 2010 season, although it accounts for very little leatherback nesting (Lagueux and Campbell unpublished data; Lagueux *et al.* 2012)." Leatherback nesting is not reported elsewhere on the Nicaragua Caribbean coast; however, leatherback hatchlings have been reported on the beach just north of the Río Grande de Matagalpa river mouth, although their origin is not known (Lagueux and Campbell unpublished data). Threats to leatherbacks in the region include egg poaching,unintended capture in entanglement nets set for green turtles and gill nets, and direct harvest of nesting of females (Lagueux and Campbell unpublished data; Lagueux *et al.* 2005).

A small amount of nesting also occurs in Honduras (Lagueux and Campbell 2005). In the past 10 years, an increasing number of projects have been initiated to monitor leatherbacks in this region. No trend analyses are available in the literature.

In Costa Rica, Tortuguero, leatherback nesting has decreased 88.5% overall from 1995 through 2011 (Gordon and Harrison 2012). Troëng *et al*. (2007) estimated a 67.8% overall decline from 1995 through 2006. However, these estimates are based on an extrapolation (see Troëng *et al*. 2004) of track survey data, which has consistently underestimated the number of nests reported during the surveys (Gordon and Harrison 2012). Regardless of the method used to derive the estimate, the number of nests observed over the last 17 years has declined. From 2007 through 2011, approximately 281 nests are laid per season (Gordon and Harrison 2012). Troëng *et al*. (2004) found a slight decline in the number of nests at Gandoca, at the southernmost end of Caribbean Costa Rica, between 1995 and 2003, but the confidence intervals were large. Data between 1990 and 2004 at Gandoca averaged 582.9 ($\pm$ 303.3) nests each year, indicating nest numbers have been lower since 2000 (Chacón and Eckert 2007), and the numbers are not increasing (Turtle Expert Working Group 2007). During the 2012 nesting season, 288 leatherback nests were observed at Gandoca and a total of 4,363 nests were recorded in Pacuare, Pacuare Reserve, Estación Las Tortugas, Parismina, and Cahuita, Costa Rica (Fonseca and Chacón 2012). Other than Tortuguero and Gandoca, Costa Rica, no trend analyses are available in the literature.

In Panama, Chiriqui Beach, 1,000-4,999 nests were laid each year between 2004 and 2011 (Meyland *et. al.* 2013). An estimated 3,077 nests and 234 individual leatherbacks were identified on surveys during the 2003 and 2004 nesting seasons (Ordoñez *et al*. 2007). During 2001 through 2003, Troëng *et al*. (2004) reported that 5,759-12,893 leatherback nests were deposited annually between the San Juan River mouth (border between Costa Rica and Nicaragua) through Chiriqui Beach, Panama. Patiño-Martinez *et al.* (2008) surveyed the coast of Armila in southeastern Panama adjacent to the border with Columbia. For the 2006 and 2007 nesting seasons, approximately 897 nests km^{-1} (4,036 and 3,599 nests in 4.5 km) were estimated to have been laid, which is a greater nesting density than Chiriqui Beach, Gandoca, Pacuarue, and Tótuguero (Patiño-Martinez *et al.* 2008). In addition to surveying southeastern Panama, Patiño-Martinez *et al.* (2008) surveyed five sites through the Gulf of Urabá in Colombia. For the entire 100 km of coast surveyed from southeastern Panama through Colombia, 5,689 to 6,470 nests were estimated for 2006 and 2007, respectively. Three stretches of beach totaling 18.9 km held over 98.5-98.7% of the nesting activity in the region surveyed in the two years. Earlier studies in the Gulf of Urabá, Colombia, recorded 162 nests on a 3-km beach during the 1998 season (Duque *et al.* 2000), and an average 218 nests were laid on a 3-km beach between 1998 and 2005 on La Playona, Colombia (Patiño-Martínez *et al*. 2006). Nesting has been recorded at other beaches in Colombia, but at low numbers (e.g., Borrero Avellaneda *et al.* 2013). No trend analyses are available in the literature.

Nesting in the Southern Caribbean occurs in Venezuela, Dominica, Trinidad, Guyana, Suriname, and French Guiana. Leatherback studies in the Guianas began in the 1960s, and there is very little mention of leatherback nesting prior to this period in the literature. No trend analyses are available in the literature.

In Venezuela, 31 females were observed and 74 nests counted between March and August 2001 at Playa Parguito on Margarita Island; no previously published information exists for this beach (Hernández *et al*. 2007). Over 200 nests were reported from other parts of Venezuela in 2004 (Mast 2005-2006). From 2000 to 2009, approximately 20 to 30 females nested on Cipara and Querepare beaches, Venezuela, but monitoring effort varied between years (Velásquez *et al.* 2010). No trend analyses are available in the literature.

In Dominica, the three most important leatherback beaches were patrolled from 22 April-15 December in 2003, from 1 March-30 October in 2004, and from 17 March-30 September in

2005. Seven leatherbacks were encountered and tagged in 2003, 18 in 2004, and 12 in 2005 (Byrne and Eckert 2006; Franklin *et al.* 2004). No trend analyses are available in the literature.

Trinidad supports an estimated 7,000 to 12,000 leatherbacks nesting annually (S. Eckert unpublished data cited in Stewart *et. al.* 2013), which represents more than 80% of the nesting in the insular Caribbean Sea (Fournillier and Eckert 1999). The more recent estimate of females nesting annually is an increase from nesting seasons 2000 through 2004 in which 2,728 (1,949-3,410) were estimated to nest each year (Livingstone and Downie 2005). Data on the number of observed nests at Matura Beach in Trinidad (adjusted for number of nesting females) from 1994 to 1999, as well as the actual number of nesting female counts based on tag information for 2000-2005 (excluding 2002), indicate a positive trend over the time period. The probability that the annual growth rate exceeded 1 was 0.81 for the period between 1994 and 1999, suggesting the population was likely increasing for the duration of the time series (Turtle Expert Working Group 2007).

Leatherback work in Guyana began in 1965; however, because of the shifting nature of beaches in the region and because of varying sampling methods, data collection has not been consistent among years. Nevertheless, estimates of nest counts are available. Between 2007 and 2010, nests counts ranged from 377 to 1,762 (De Freitas and Pritchard 2008, 2009, 2010; Kalamandeen *et al.* 2007). The population may be increasing (Turtle Expert Working Group 2007).

For Suriname and French Guiana, historical estimates of the number of females nesting each year range from approximately 5,000 to 20,000 (see Fossette *et al.* 2008). Suriname and French Guiana may represent over 40% of the world's leatherback population, although the magnitude of the West African rookery needs to be verified (Spotila *et al*. 1996). In Suriname, daily nest counts have been conducted since 1969 with varying methodology over the years, and possibly less survey effort in recent years. Hilterman and Goverse (2007) identified 8,462 individual leatherbacks nesting in Suriname between 1999 and 2005. Their estimate of the minimum annual nesting number was between 1,545 and 5,500 females in Suriname. Nesting in French Guiana has been cyclic with nesting varying between approximately 5,029 and 63,294 nests annually between 1967 and 2005 (Turtle Expert Working Group 2007). Rivalan *et al*. (2006) estimated a population of 2,750-20,000 individuals (males and females of all life stages) from the Maroni (Suriname and French Guiana). They determined that 90-220 individuals were needed to maintain adequate genetic variance for adaptive evolution ("effective population size"). Girondot *et al*. (2007) analyzed nesting data collected between 1967 and 2002 from French Guiana and Suriname and found that the population can be classified as stable or slightly increasing. The Turtle Expert Working Group (2007) analyzed nest numbers from 1967-2005 and found a positive population growth rate over the 39-year period for French Guiana and Suriname. Fossette *et al.* (2008) excluded the data prior to 1977 due to poor or unknown survey quality and added the data for 2003-2005. They found a slight, but not significantly different from zero, growth rate, which they interpreted as a stable population during the period (Fossette *et al.* 2008). For the purposes of this 5-year review, we accept the more recent analysis (Fossette *et al.* 2008) because it excludes poor data.

In Brazil, 527 nests were reported between 1988-1989 and 2003-2004 although annual numbers varied between 6 in 1993-1994 and 92 in 2002-2003 (Thomé *et al*. 2007). A 20.4% increase in nesting was observed on average annually between 1995-1996 and 2003-2004 (Thomé *et al*. 2007). Analyses of data between 1988 and 2003 found an estimated annual growth rate of 1.08 with the estimated 95% confidence interval of 1.04-1.13; the probability of the population increasing was greater than 0.99 (Turtle Expert Working Group 2007).

In West Africa, nesting has been reported in Mauritania, Senegal, the Bijagos Archipelago of Guinea-Bissau, Turtle Islands and Sherbro Island of Sierra Leone, Liberia, Cote d'Ivoire, Ghana, Togo, Benin, Nigeria, Cameroon, Sao Tome and Principe, Bioko and continental Equatorial Guinea, Islands of Corisco in the Gulf of Guinea, Angola, and the Democratic Republic of the Congo (summarized in Fretey *et al*. 2007a). The number of leatherback nests recorded on Bioko (Equatorial Guinea) for 1996/1997 was 862 and 1,170 in 1997/1998 (Tomás *et al.* 2000). The mean number of nests recorded on five nesting beaches of Bioko in 2000/2001 and 2004/2005 was 3,896 (2,127-5,071) (Rader *et al*. 2006). In the Republic of the Congo, 70 leatherback nests were counted between October and December during the 2003/2004 nesting season along 20 km of beach (Renatura Report 2004), and at least 148 nests were recorded during the 2005/2006 nesting season (Renatura Report 2006). Long-term surveys are needed to understand overall nesting trends in the Republic of the Congo (Girard and Breheret 2013).

The most important nesting beaches for leatherbacks in the eastern Atlantic Ocean are in Gabon where the total breeding population is estimated between 15,730 to 41,373 females that primarily nest in protected areas (Witt *et al.* 2009). Billes *et al*. (2000) estimated 30,000 nests along 96.5 km of Mayumba Beach in southern Gabon during the 1999/2000 nesting season. Billes *et al*. (2003) estimated that 6,300 females had nested during the 1999/2000 season and 7,800 females during the 2000/2001 season, although it should be noted that the number of females nesting may be a slight overestimate as clutch frequency may have been underestimated. Nest density at Kingere in the Pongara National Park, Gabon, is estimated to be 170 to 450 nests per kilometer (Ikaran 2013). A 5.75 km stretch of beach at Gamba, Gabon, was surveyed from 2002 through 2007 where the number of nests fluctuated with highs and lows each year ranging from 128 to 851 nests (Mounguéngui 2007). Data from 2006-2007 collected by the Gabon Sea Turtle Partnership (unpublished data) indicate figures consistent with the 1999-2000 nesting season. However, a steep decline had been noted during the 2003-2004 and 2004-2005 seasons. These estimates highlight the importance of this region and the need to re-evaluate global population size in light of the greater contribution from West Africa. Given the short-term nature of nesting surveys in West Africa, it is not possible to conduct any trend analyses for any of the populations in this region. Interestingly, similar fluctuations have been documented for other nesting beaches in the western Atlantic Ocean, such as Gondoca Beach (Chacón-Chaverri and Eckert 2007) and Tortuguero, Costa Rica (Troëng *et al*. 2004).

Nesting has not been recorded in South Africa along the Atlantic coast. However, adults from South Africa in the Indian Ocean nesting population have migrated around the Cape of Good Hope into southeast Atlantic waters (Hughes *et al.* 1998; Luschi *et al.* 2003b, 2006).

Indian Ocean and Southeast Asia

Hamann *et al*. (2006a) conducted a thorough assessment of leatherbacks in all the countries of the Indian Ocean and Southeast Asia region and identified the following four leatherback nesting sub-regions that may qualify as separate management units due to several factors, including possible independent breeding assemblages and differences in management regimes. Western Pacific countries and Malaysia were included within the assessment of the Indian Ocean and Southeast Asia (Hamann *et al*. 2006a). Although their synthesis is presented here, these countries will be revisited in greater detail under the Pacific Ocean section. Wallace *et al.* (2010b) also conducted a global assessment of leatherback populations and found that the northeast Indian Ocean (63 nesting sites) could be grouped as a separate regional management unit from the southwest Indian Ocean (8 nesting sites) based on biogeography.

South African index beaches demonstrated an increase from 10-20 nesting females annually in the 1960s to approximately 100 females annually in the 1990s (Hughes 1996). Nesting data between 1963 and 1997, indicate an estimated annual growth rate of 1.04 with estimated 95% posterior interval of approximately 1.03-1.05 (Turtle Expert Working Group 2007). Maputaland and St. Lucia Marine Reserves, South Africa, have been monitored since 1965 (Nel *et al.* 2013). Nesting data from 1965 to 2010 collected from a 12.8 km index area and expanded in 1973 to a total 56 km show an oscillating, but stable, nesting trend for the index area. However, the nesting trend for the entire 56 km area shows a decline since 1994 (Nel *et al.* 2013).

In Mozambique, nesting numbers have not been well recorded, but estimates between 1994 and 2004 suggest that approximately 10 females nest per year in southern Mozambique -- no increase in nesting has been observed in Mozambique (Hamann *et al*. 2006a; Nel 2012).

Long-term data sets are not common in the countries of the northeastern Indian Ocean (Bay of Bengal, Sri Lanka, Andaman and Nicobar Islands), southern Indonesia, and Australia. Hamann *et al*. (2006a) summarized the following information from this sub-region: in Sri Lanka the nesting population may consist of 100 to 200 females annually (based on a year of data), whereas in the Andaman and Nicobar Islands there are approximately 400 to 600 females nesting per year. Thailand supports about 10 nests per year. In Java, opposite trends are observed in two neighboring rookeries -- Meru Betiri, nesting numbers have declined from 20 females/year in the 1980s to less than five females/year in the early 2000s, whereas at neighboring Alas Perwo the nesting population possibly could have doubled over the same period of time (from 500 eggs annually to 1,000 eggs annually) although clutch frequency data for Alas Perwo are not available. Arnhem Island, Australia, has not been completely surveyed and known nesting is irregular.

For the remaining potential nesting areas in the Indian Ocean and Southeast Asia, some nesting has been suspected in the past in Bangladesh, Myanmar, and Somalia, but there are no current nesting accounts (Hamann *et al*. 2006a). A systematic survey of a 12 km of nesting beach on Sonadia Island, Bangladesh, was conducted from 2005-2010, and no leatherback nesting activity was reported (Islam *et al.* 2011). Historically, fishermen have reported leatherbacks nesting on the mainland coast of India, but in low numbers (Pandit and Soans 2013; Shanker 2013). Only 15 leatherbacks, mostly stranded or caught in fisheries gear, have been recorded between 1923 and 2009 (Shanker 2013). In Kenya, no nesting has been recorded, but only 50% of the coastline has been surveyed. Rare and anecdotal accounts of leatherback nesting exist in the Philippines. For example, a leatherback was reported to nest near Legazpi City in the Albay Gulf, Philippines in July 2013 (Arguelles 2013), and leatherbacks have been recorded to strand along the coast of Mindanao, Philippines, indicating a presence in coastal waters (Lucero *et al.* 2011). In the

Seychelles, there have been two unconfirmed reports of leatherback nesting, but no current reports. Only one leatherback nest has been reported from the Chagos Islands in the 1970s, although since then surveys have not found any evidence of leatherback nesting. Community surveys in Vietnam indicate that nesting numbers have dropped from 500 females per year (= thousands of nests) prior to the 1960s to less than 10 nests annually today (Hamann *et al*. 2006a, 2006b). Only two leatherback clutches have been recorded in Japan after extensive surveys.

Pacific Ocean

A dramatic drop in nesting numbers has been recorded on major nesting beaches in the Pacific Ocean, although a sizeable, but severely declining, nesting population exists in Papua, Indonesia (Dutton *et al.* 2007; Hitipeuw *et al.* 2007; Tapilatu *et al.* 2013). Spotila *et al*. (2000) highlighted the dramatic and possible extirpation of leatherbacks from key nesting beaches in the eastern Pacific.

In the western Pacific, the major nesting beaches in Papua, Indonesia, Papua New Guinea, Solomon Islands, and Vanuatu (Limpus 2002, Dutton *et al*. 2007) consist of approximately 2,700-4,500 breeding females. This number is substantially higher than the population estimate of 1,775 to 1,900 western Pacific breeding females published in 2000 and used to predict possible extinction in the Pacific (Spotila *et al.* 2000). However, this estimate should be interpreted with caution because it was derived from nest counts, and reliable data on the number of nests per female are not available (Dutton *et al*. 2007). The current overall estimate for Papua Barat, Indonesia, Papua New Guinea, and Solomon Islands is 5,000 to 10,000 nests per year (Nel 2012).

In Papua, Indonesia, the main nesting beaches occur in the northwest, also known as the Bird's Head Peninsula, Jamursba-Medi and Wermon, where approximately 75% of regional nesting occurs (Hitipieuw *et al*. 2007). Nesting numbers have dropped from over 14,000 nests recorded in 1984 at Jamursba-Medi (Bhaskar 1985) to 1,596 in 2011 (Tapilatu *et al.* 2013). From 2005 through 2011, the Jamursba-Medi nesting population has declined 52% and the Wermon population by 62.8% (Tapilatu *et al.* 2013). This represents an alarming 5.9% decrease each year resulting in only about 500 females (based on a mean clutch frequency of 5.5±1.6) nesting each year in Jamursba-Medi and Wermon at what is likely the "last remaining stronghold for leatherbacks in the Pacific"(Tapilatu *et al.* 2013). In the Manokwari region, West Papua, Indonesia, nesting occurs year round and the number of nests recorded from 2008 through 2011 ranged from 84 to 135, however survey effort was limited and likely not consistent across years (Suganuma *et al.* 2012).

Papua New Guinea hosts approximately 20% of western Pacific leatherback nesting activity, which occurs predominately along the Huon Gulf coast (Dutton *et al.* 2007). Long-term nesting trends are difficult to determine given changes in monitoring effort since 2000 (Pilcher 2012). In 2004, an aerial survey counted 415 nests along the 4,516 km flown, with 71% of nests within the Huon Gulf coast (Benson *et al*. (2007a). For a 2-km stretch at Kamiali, Papua New Guinea, Benson *et al*. (2007a) reported that between 2000/2001 and 2003/2004 the total number of females estimated to have nested ranged from 41 to 71. Ground surveys at Kamiali recorded a total of approximately 215 nesting events for 1999/2000 through 2003/2004 (Benson *et al.* 2007a). During the 2010-2011 nesting season along the Huon Coast, 79 leatherbacks nested laying a total of 527 nests (Pilcher 2011b). Of the 211 nests laid between October 2012 and February 2013, 22% were lost to erosion, poaching or did not hatch (Pilcher 2013). Although the population fluctuates each year, it appears generally stable since the 2006-2007 nesting season, when monitoring effort was standardized. But overall nest counts have declined approximately 93% since 1980 estimates when approximately 300 females were estimated to nest annually (Bedding and Lockhart 1989; Hirth *et al.* 1993; Pilcher 2009) suggesting a decline in nesting numbers. For example, Hirth *et al.* (1993) recorded 76 nests at Labu Tali during a 15 day period in 1989 over a distance of 725 meters of beach. In 2010, Pilcher (2011a) recorded 59 nests at Labu Tali over a distance of 3.2 km of beach during a 218 day period. Given the greater spatial and temporal survey conducted in 2010, the nesting numbers at Labu Tali likely represent a decline in nesting activity. While these results may be an artifact of sampling inconsistencies, it is entirely possible that the leatherback population in Papua New Guinea experienced near total nest harvest for some 40+ years through egg collection after World War II until conservation efforts were first implemented in 2000 (Bedding and Lockhart 1989; Bellagio Steering Committee 2008; Hirth *et al.* 1993; Quinn *et al.* 1983). In the Autonomous Region of Bougainville Island, Papua New Guinea, aerial surveys conducted in during the 2006-2007 nesting seasons, estimated 160-415 nests were laid per year (Dutton *et al.* 2007). Ground surveys conducted in 2009, found only 46 leatherback nests with a high level (83-100%) of nest harvest and relatively frequent harvest of adult leatherback turtles (Kinch *et al.* 2012).

In the Solomon Islands, nesting 30 years ago occurred at more than15 beaches (Vaughan 1981). Dutton *et al.* (2007) estimated that approximately 640 - 700 nests were laid annually in the Solomon Islands in 1999 – 2006 representing approximately 8% of the total western Pacific leatherback metapopulation at that time. Important nesting areas remain on Isabel Island at two principal beaches, Sasakolo and Litogarhira, with additional nesting occurring on Rendova and Tetepare in the Western Province (Dutton *et al.* 2007). Nesting beach monitoring began in 1993 at Sasokolo by the Department of Fisheries where an average of 25 females deposit approximately 100 nests per season (Pita and Broderick 2005; Ramohia *et al.* 2001). The Tetepare Descendants' Association turtle monitoring program has operated since 2002 supporting beach rangers to monitor nesting activity at Tetapare and Rendova and has permanently closed a 13 km beach to harvest. At Tetapare, approximately 30-50 leatherback nests are laid seasonally (Goby *et al.* 2010; MacKay 2005). At Rendova, 79 nests were laid during the 2009-10 winter nesting season of which only three hatched (Goby *et al.* 2010), and during the 2003-04 winter nesting season, 235 leatherback turtle nests were recorded of which only 14 hatched (Pilcher 2010), strongly suggesting that low hatch success poses significant impact to the current nesting population in the Solomons. There is no long-term data to assess trends in the Solomon Islands, although the number of nesting females is estimated to be around 100 per year (Petro *et al* 2007).

In Vanuatu, recent nesting beach surveys and a review of leatherbacks by Petro *et al*. (2007) indicate that the small nesting populations of leatherbacks on these islands have declined significantly. There appears to be low levels of scattered nesting on at least four or five beaches with a total of approximately 50 nests laid per year (Dutton *et al.* 2007). Leatherbacks nest in small numbers on many of the islands, but approximately 10-15 females nest at the primary nesting beach at Votlo on Epi Island where surveys have been conducted since 2002/03. During the 2010/11 nesting season, 41 nests were laid at Votlo, although only 8 nests hatched (Petro 2011). Other potentially good nesting sites need to be thoroughly surveyed.

In Malaysia, the major nesting rookery at Rantau Bang in Terengganu has collapsed from over 10,000 nests in the 1950s to 10 or fewer nests in recent years (reviewed by Eckert *et al.* 2012). The decline is attributed to fisheries interactions and egg harvest (reviewed by Eckert *et al.* 2012).

In Fiji, 20-30 individuals are estimated to nest each year (Rupeni *et al*. 2002).

In southeastern Australia, nesting is sporadic with less than a handful of nests each year (Dobbs 2002). Wreck Rock Beach in southern Queensland, Australia, reported an average 0-3 nests annually from 1969 to 1995, but no nests have been observed since 1995, despite regular monitoring (Flint *et al.* 2012). Nesting is irregular in northern Australia (Hamann *et al*. 2006a).

In the eastern Pacific, major nesting beaches are found in Costa Rica, Mexico, and Nicaragua. At Las Baulas National Marine Park, Costa Rica, which consists of Playa Grande and the smaller nesting beaches of Playa Langosta and Playa Ventanas, Santidrián Tomillo *et al*. (2007) analyzed data for the area and reported that leatherback numbers declined over 15 years of monitoring (1988-1989 to 2003-2004) with approximately 1,504 females nesting in 1988-1989 to an average of 188 females nesting in 2000-2001 and 2003-2004. An earlier estimate of nesting females at just Playa Grande showed a steady drop from 1,367 females in 1988-1989 to 506 in 1994-1995, and down to 117 by 1998-1999 (Spotila *et al*. 2000).

In Pacific Mexico, Pritchard (1982) conducted an aerial survey of the coastline and derived an estimate of several thousands of nesting females. Although nesting occurs at many sites along the coast (e.g., Tomatal Beach: Vannini and Rosales Jaillet 2009), monitoring on four primary index beaches (Mexiquillo, Tierra Colorada, Cahuitán, Barra de la Cruz) for over 20 years (1982-2004) has shown a decline in nest numbers. Tens of thousands of nests were likely laid on the beaches in the 1980s, but during the 2003-2004 season a total of 120 nests was recorded on the four primary index beaches combined (Sarti Martinez *et al*. 2007).

In Pacific Nicaragua, a monitoring and protection program was established in 2002 at Veracruz beach, followed by Juan Venado in 2004, and Salamina in 2008 (Urteaga *et al.* 2012). From 2002 to 2010, 420 nests were recorded and 48 individual females were identified. Nesting numbers have decreased since 2006 (Urteaga *et al.* 2012). However, the monitoring period is too short to determine a population trend.

Mediterranean Sea

Leatherbacks are found in the Mediterranean Sea. However, nesting in the region is not known or is believed to be extremely rare (Camiñas 1998; reviewed by Casale and Margaritoulis 2010).

Table 1. Leatherback nesting population site location information where multiple-year surveys were conducted or trends are known (data type, years surveyed, annual number (nests, females, trend). Nesting population trend symbols: ▲ = increasing; ▼ = decreasing; ▬ = stable; ? = unknown. See text in 'Abundance and Population Trends' for greater detail.

Location	Data: Nests, Females	Years	Annual Number	Trend	Reference
ATLANTIC					
United States (Florida)	Nests	1979 - 2008	63-754	▲	Stewart *et al.* 2011a
Puerto Rico (Culebra)	Nests	1993 - 2012	395-32	▼	C. Diez, Department of Natural and Environmental Resources of Puerto Rico,, unpublished data; Diez *et al.* 2010; Ramírez-Gallego *et al.* 2013
Puerto Rico (other)	Nests	1993 - 2012	131-1,291	▲	C. Diez, Department of Natural and Environmental Resources of Puerto Rico, unpublished data
United States Virgin Islands (Sandy Point National Wildlife Refuge, St. Croix)	Nests	1986 - 2004	143-1,008	▲[1]	Dutton *et. al.* 2005; Turtle Expert Working Group 2007
British Virgin Islands	Nests	1986 - 2006	0-65	▲	McGowan *et al.* 2008; Turtle Expert Working Group 2007
Nicaragua	Nests	2008 - 2013	42-132	?[2]	C. Laguex and C. Campbell, Wildlfie Conservation Society, unpublished data
Costa Rica (Tortuguero)	Nests	2007 - 2011	~281	▼	Gordon and Harrison 2012
Costa Rica (Gandoca)	Nests	1990 - 2004	~583	▼	Chacón and Eckert 2007; Turtle Expert Working Group 2007
Panama (Chiriqui Beach)	Nests	2004 - 2011	1,000-4,999	?	Meylan *et al.* 2013
Colombia	Nests	2006 - 2007	1,653-2,871	?	Patino-Martinez *et al.* 2008
Trinidad	Females	1994 - 2005	2,096	▲	Turtle Expert Working Group 2007
Guyana	Nests	2007 - 2010	377-1,722	▲	De Freitas and Pritchard 2008, 2009, 2010; Kalamandeen *et al.* 2007; Turtle Expert Working Group 2007
French Guiana	Nests		5,029-63,294	▬	Fossette *et al.* 2008
Suriname	Nests		2,732-31,000	▬	Fossette *et al.* 2008
Brazil	Nests	1988 - 2004	6-527	▲	Thome *et al.* 2007; Turtle Expert Working Group 2007
Equatorial Guinea (Bioko)	Nests	2000 - 2005	2,127-5,071	?	Rader *et al.* 2006
Congo	Nests	2003 - 2006	70-148	?	Rentaura Report, 2004, 2006

Location	Data: Nests, Females	Years	Annual Number	Trend	Reference
Gabon	Nests	2002 - 2007	36,185-126,480	?	Witt *et al.* 2009
INDIAN OCEAN					
South Africa	Nests	1965 - 2010	~296	▬[3]	Nel *et al.* 2013
Mozambique	Females	1994 - 2004	~10	?	Hamann *et al.* 2006a
PACIFIC					
Indonesia (Papua-Jamursba-Medi)	Nests	1984 - 2011	14,522-1,596	▼	Tapilatu *et al.* 2013
Indonesia (Papua-Wermon)	Nests	2002 - 2011	2,994-1,096	▼	Tapilatu *et al.* 2013
Papua New Guinea (Labu Tali)	Nests	1989 - 2011	76–59	▼[4]	Hirth *et al.* 1993; Pilcher 2011a
Vanuatu	Nests	2002 - 2010	~50	▼	Petro 2011; Petro *et al.* 2007
Malaysia (Terengganu)	Nests	1956 - 2009	10,000-10	▼	reviewed by Eckert *et al.* 2012
Costa Rica (Las Baulas National Marine Park: Playa Grande, Langosta, and Ventanas)	Females	1988 - 2004	1,504-188	▼	Santidrián Tomillo *et al.* 2007
Mexico (Mexiquillo, Tierra Colorada, Cahuitán, Barra de la Cruz)	Nests	1982 - 2004	>10,000-120	▼	Sarti Martinez *et al.* 2007
Nicaragua (Veracruz, Juan Venado, and Salamina)	Nests	2002 - 2010	~53	?	Urteaga *et al.* 2012

[1] A more recent trend analysis was not found in the literature. However, trends since 2001 suggest the population may be declining, possibly due to a decrease in the number of new nesters, lowered productivity (number of clutches per season and lower hatch success), and an increase in remigration intervals (Garner 2012; Garner and Garner 2010; Garner *et al.* 2012).

[2] The number of nests likely underrepresents the area because 22% of nesting activity was not surveyed from 2011-2013 due to military presence (Laguex and Campbell, Wildlife Conservation Socitey, unpublished data).

[3] Based on 12.8 km index area in Maputaland and St. Lucia Marine Reserves, South Africa.

[4] Survey distance and time differed between the two surveys at Labu Tali, but the weight of evidence from the area indicates a declining population.

Summary

Leatherback nesting populations are declining dramatically in the Pacific Ocean, yet appear stable in many nesting areas of the Atlantic Ocean and South Africa in the Indian Ocean. Many ideas have been provided to explain the disparate population trends seen in the Pacific, Atlantic, and Indian Oceans (reviewed by Saba 2013; Wallace and Saba 2009).

Overall, leatherback females in the Pacific Ocean, particularly those originating from the eastern Pacific, are smaller and less productive than females in the Atlantic Ocean and Indian Ocean (i.e., South Africa). Adult size and fecundity may affect population trends, and studies largely attribute greater variability in resource abundance and distribution in the eastern Pacific Ocean compared to the Atlantic Ocean and southeastern Indian Ocean to the differences in ocean basin population trends (reviewed by Saba 2013; Wallace and Saba 2009). For example, mean primary productivity in all the foraging areas used by the western Atlantic females is significantly higher (150% greater) than those of the eastern Pacific females; the reproductive output of western Atlantic females was double that of eastern Pacific females (Saba *et al*. (2008a). Another possible measure of the difference between the productivity of the Atlantic Ocean and Pacific Ocean is the amount of prey ingested by foraging adults. Adult leatherbacks feeding in the highly productive areas of the western Atlantic Ocean are estimated to consume 73% of their body mass daily to meet their energetic demands (Heaslip *et al.* 2012); whereas, in the Pacific Ocean, they consume 26% of their body mass daily (Jones *et al.* 2012). Wallace *et al*. (2006a) calculated reproductive energy budgets for leatherbacks in the North Atlantic (St. Croix) and the eastern Pacific (Costa Rica). They found that resource limits in the eastern Pacific due to the El Niño Southern Oscillation (ENSO; which is an irregular pattern of periodic variation between warm and cool sea surface temperatures) may cause longer remigration intervals, thereby lowering reproductive success and increasing exposure to fisheries. Saba *et al*. (2007, 2008a, 2008b) evaluated the effect of ENSO on the reproductive frequency of eastern Pacific leatherback turtles and reported that declines are not only due to fisheries but also due to sensitivity to the interannual climate variability, governed by ENSO, which is reflected in their remigration probabilities (higher remigration probability was seen during La Niña years than El Niño years). Except for the eastern Pacific, which had interannual variability in primary production because of ENSO, all of the foraging areas had seasonal primary production. High reproductive output and consistent and high quality foraging areas in the Atlantic have contributed to the stable or recovering populations in the Atlantic Ocean (Saba *et al*. 2008a).

Foraging strategies may contribute to the observed decline in the Pacific. As discussed earlier, the large population in the western Pacific has multiple nearshore and pelagic foraging areas in the northern and southern hemispheres (Benson *et al.* 2011). The eastern Pacific population, however, has a very narrow foraging strategy (by feeding primarily in the southeastern Pacific, thereby making it more susceptible to negative anthropogenic impacts and climatic stochasticity. The western Pacific population, in contrast, is better buffered against such perturbations (Dutton 2006). Leatherback foraging patterns also differ between the Pacific and Atlantic Ocean populations (Bailey *et al.* 2008, 2012). In the Pacific Ocean, leatherbacks spent long periods transiting over widely dispersed areas, indicating food availability was patchy (Bailey *et al.* 2008, 2012). By contrast, in the Atlantic Ocean, leatherback transit was frequently interspersed with foraging behavior, indicating leatherbacks frequently encountered areas of high prey density (Bailey *et al.* 2012). Prey abundance and distribution may explain why the Pacific populations are in worse shape than the Atlantic populations (Bailey *et al.* 2012). Finally, many nesting beaches in the Pacific are extremely remote, which may hinder sustainable conservation efforts seen at the more accessible beaches in the Atlantic.

2.3.2 Five-Factor Analysis (threats, conservation measures, and regulatory mechanisms)

The determination to list a species under the ESA is based on the best scientific and commercial data regarding the five listing factors (see below). Subsequent 5-year reviews completed in accordance with section 4(c)(2) of the ESA must also make determinations about the listing status based, in part, on these same factors.

2.3.2.1 Present or threatened destruction, modification or curtailment of its habitat or range:

Leatherbacks are increasingly threatened by natural and anthropogenic impacts to their nesting beaches and coastal and pelagic marine habitat. Natural factors, including the 2004 tsunami in the Indian Ocean (see detailed report by Hamann *et al*. 2006c) and the tsunami that affected Japan in 2011, may have impacted leatherback nesting beach habitat through encroachment and erosion (2004 tsunami) or may have resulted in increased debris into leatherback marine habitat (e.g., impacting migratory routes and foraging hotspots). Shifting mudflats in the Guianas have also made nesting habitat unsuitable (Crossland 2003, Goverse and Hilterman 2003). Human activities also impact leatherback habitat, including development and tourism in several countries, which affect nesting beaches and adjacent waters (e.g., Hamann *et al*. 2006a; Hernández *et al*. 2007; Santidrian Tomillo *et al*. 2007; Maison 2006; Mangubhai *et al.* 2012). Impacts to nesting habitat include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Bouchard *et al*. 1998; Lutcavage *et al*. 1997).

In Gabon, inland timber logging poses a major threat to nesting leatherbacks (Ikaran 2013; Laurance *et al.* 2008; Pikesley *et al.* 2013). Harvested timber are transported by rivers or sea to major ports and many are lost or intentionally released and end up on the nesting beaches of Gabon (Laurance *et al.* 2008). Aerial surveys conducted in in 2003, 2007, and 2011 recorded between 15,160 and 17,262 beached logs along 550 km of coastline, with the highest density (212 logs km^{-1}) reported for Pongara National Park, Gabon (Pikesley *et al.* 2013). During the 2006/2007 nesting season at Pongara National Park, 15 female leatherbacks were found dead among the logs and another 28 were rescued as they had become dehydrated and were unable to reach the sea unaided (Ikaran 2013). Daily counts of nesting activity and impacts associated with beached logs across the 2006/2007 to 2010/2011 nesting seasons showed an average 17% of nesting females were impacted by the debris while nesting at Pongara National Park (Pikesley *et al.* 2013).

Accumulation of marine debris on the beach, as well as sand mining, can have a negative impact on available nesting habitat in some areas (Chacón-Chaverri 1999, Formia *et al*. 2003). 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; Witherington *et al*. 2003, 2007). Coastal development is usually accompanied by artificial lighting, and the presence of lights on or adjacent to nesting beaches alters the behavior of nesting females and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Bourgeois *et al.* 2009; Cowan *et al*. 2002; Deem *et al*. 2007;

Witherington 1992; Witherington and Bjorndal 1991). In many countries, coastal development and artificial lighting are responsible for substantial hatchling mortality. Although outreach and conservation programs controlling these impacts does exist in countries such as Costa Rica, Mexico, and the United States (Lutcavage *et al*. 1997), a majority of countries do not have regulations in place. Fortunately, some of the major nesting beaches occur in sufficiently remote areas, and large-scale development is less of an issue there. In the western Pacific, leatherbacks face similar issues as those described above from timber harvest, village sprawl, beach erosion, and predation of nests by dogs and pigs (Bellagio Steering Committee 2008; Hitipeuw *et al.* 2007; Pilcher 2009; Tapilatu *et al.* 2013).

Considering that coastal development and beach armoring is detrimental to leatherback nesting behavior (Lutcavage *et al*. 1997), human population expansion is reason for major concern. This is underscored by the fact that over the next few decades the human population is expected to grow by more than 3 billion people (about 50%). By the year 2025, the United Nations Educational, Scientific and Cultural Organization (UNESCO) (2001) forecasts that population growth and migration will result in a situation in which 75% of the world human population will live within 60 km of the sea. Such a migration undoubtedly will change a coastal landscape that, in many areas, is already suffering from human impacts. The problems associated with development in these zones will progressively become a greater challenge for conservation efforts, particularly in the developing world where wildlife conservation can be secondary to other national needs.

As leatherbacks forage widely in the oceanic habitat, modifications to foraging areas are more difficult to monitor. For example, their marine (and nesting) environment is impacted by the petroleum industry. Numerous oil platforms operate off Gabon. Billes and Fretey (2004) found debris and tar balls that likely came from these operations. Oil spills are a concern. In 2010, a major oil spill occurred in the north central U.S. Gulf of Mexico, affecting important foraging habitat used by leatherbacks (Evans *et al.* 2012; Witherington *et al.* 2012). Assessment of the harm is ongoing as part of the Natural Resources Damage Assessment.

In the marine environment, marine debris may also serve as a source of mortality to all species of sea turtles, as small debris can be ingested and larger debris can entangle animals, leading to death. Manmade materials such as plastics, micro plastics, and derelict fishing gear (e.g., ghost nets) that may impact leatherbacks via ingestion or entanglement can reduce food intake and digestive capacity, cause distress and/or drowning, expose turtles to contaminants, and in some cases cause direct mortality (Arthur *et al*. 2009; Balazs 1985; Bjorndal *et al*. 1994; Doyle *et al.* 2011; Keller *et al.* 2004; Parker *et al*. 2011; Wabnitz and Nichols 2010). While the impact of marine debris on leatherbacks during their pelagic life stage is currently unquantified, it is likely that impacts may be severe, given the increase of plastics and other debris and pollution entering the marine environment over the past 20-30 years.

Impacts from climate change, especially due to global warming, are likely to become more apparent in future years (Intergovernmental Panel on Climate Change (IPCC) 2007a). Based on the available information, climate change is an anthropogenic factor that will affect leatherback habitat and biology. The global mean temperature has risen 0.76ºC over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (IPCC 2007a).

Levels of atmospheric carbon dioxide have almost reached 400 parts per million (http://www.esrl.noaa.gov/gmd/ccgg/trends/weekly.html), a level not recorded since the Pliocene Epoch. Based on substantial new evidence, observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. These changes include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007b), which could affect leatherback prey distribution and abundance. Global warming is expected to expand foraging habitats into higher latitude waters (James *et al*. 2006; McMahon and Hays 2006), and change habitat conditions on the beach (e.g., Pike 2013). See section 2.3.2.5 for further discussion on impacts due to climate change.

The Services believe that leatherbacks remain in danger of extinction because of destruction, modification, and curtailment of their habitat.

2.3.2.2 Overutilization for commercial, recreational, scientific, or educational purposes:

Egg collection occurs in many countries around the world (e.g., Billes and Fretey 2004; Bräutigam and Eckert 2006; Chan and Liew 1996; Fretey *et al*. 2007a; Hamann *et al*. 2006a, 2006b; Hilterman and Goverse 2007; Kinan 2002; Maison *et al.* 2010; Mangubhai *et al.* 2012; Santidrián Tomillo *et al*. 2007, 2008; Troëng *et al*. 2007). For example, during the 2012 nesting season, 55% (283 of 514) of leatherback nests were poached on Pacuare Playa, Costa Rica (Fonseca and Chacón 2012). Egg harvest has been attributed to catastrophic declines such as in Malaysia. Despite conservation efforts, egg harvest continues at certain levels in Indonesia, Papua New Guinea, Solomon Islands and Vanuatu (Bellagio Steering Committee 2008). On some beaches (e.g., South Africa), egg harvests are now a thing of the past (Hamann *et al*. 2006a). In the Republic of Congo, harvest of females and eggs was almost 100% in 2003, but has decreased to 5% over the past decade due to beach monitoring and community outreach (Girard and Breheret 2013). On the Pacific coast of Nicaragua, egg poaching has been dramatically reduced since a monitoring and protection program began in 2002 (Urteaga *et al.* 2012). Harvest of females remains a matter of concern on many beaches (e.g., Bräutigam and Eckert 2006; Chacón and Eckert 2007; Fretey *et al*. 2007a; Fournillier and Eckert 1999; Gomez *et al*. 2007; Hamann *et al*. 2006a; Kinch *et al.* 2012; Ordonez *et al*. 2007). A traditional harvest of subadult and adult leatherbacks occurs in the Kei Islands (Lawalata *et al*. 2006; Suarez and Starbird 1996). However, most villages in the Kei Islands have abandoned the harvest and the annual take estimate is 10-15 leatherbacks (C. Hitipeuw, World Wildlife Fund Indonesia, 2013 personal communication). Local communities are interested in developing ecotourism programs that rely on healthy leatherback populations, and there is hope that the harvest levels will remain low into the future (C. Hitipeuw, World Wildlife Fund Indonesia, 2013 personal communication). Leatherbacks are also used in voodoo ceremonies and traditional medicine in West African countries (Fretey *et al.* 2007b), as well as religious ceremonies in Taiwan (Cheng and Chen 1997).

Low hatching success is characteristic of leatherbacks despite high fertility rates (reviewed by Bell *et al*. 2003; Eckert *et al.* 2012) and when additional anthropogenic or predation pressures are placed on incubating eggs, a management strategy commonly undertaken is nest relocation. However, many studies have found that hatching success of nests relocated to another section of the beach or to hatcheries is lower than *in situ* nests (reviewed by Eckert *et al.* 2012; Hernández *et al*. 2007); although another study found adequate hatching success in relocated nests at St. Croix (Eckert and Eckert 1990), which may be a factor in the increase observed in this nesting population (Dutton *et al.* 2005). Translocating nests into hatcheries also may skew natural sex ratios. In Playa Grande, Costa Rica, fewer females were produced in translocated nests where lower hatch success may have resulted in cooler nests due to fewer eggs producing metabolic heat (Sieg *et al.* 2011). Poor hatchery practices have skewed natural sex ratios, resulting in 100% females produced in some facilities (Chan and Liew 1995). The consequences of nest relocation need to be carefully evaluated (Mrosovsky 2006).

The Services believe that leatherbacks remain in danger of extinction because of ongoing activities related to commercial, recreational, and scientific (i.e., translocation of nests) purposes.

2.3.2.3 Disease or predation:

Flint (2013) recently summarized diseases of concern for sea turtles, but the health status of and baseline blood indices for leatherbacks have been largely unstudied. Baseline health data were obtained from 19 leatherbacks (12 directly caught at-sea for satellite tagging and 7 incidentally captured in fishing gear) in the Northwest Atlantic Ocean (Innis *et al.* 2010). Although most were determined to be in good health, several leatherbacks had evidence of past injuries and entangled turtles exhibited blood values indicating stress or reduced food or seawater uptake due to entanglement (Innis *et al.* 2010). Deem *et al*. (2006) presented the first baseline values for hematology, plasma biochemistry, and plasma protein electrophoresis from 35 leatherbacks nesting in Gabon and also measured plasma corticosterone, vitamin concentrations, and several toxicological parameters; the sampled leatherbacks were rated as being in good health. The first case of fibropapillomatosis in leatherbacks was reported from Pacific Mexico (Huerta *et al*. 2002). This disease is a condition likely caused by a herpesvirus (Ene *et al*. 2005) and is characterized by the presence of internal and external tumors (fibropapillomas) that may grow large enough to hamper swimming, vision, feeding, and potential escape from predators (Herbst 1994). Fibropapillomatosis is not as common in leatherbacks as in other sea turtle species (Huerta *et al*. 2002).

Leatherbacks are preyed upon by a variety of predators (reviewed by Eckert *et al.* 2012). Predators of eggs include feral pigs and dogs, (e.g., Hamann *et al*. 2006a; Hitipeuw *et al*. 2007; Ordonez *et al*. 2007; Pilcher 2009; Tapilatu and Tiwari 2007), mole crickets (Maros *et al*. 2003), raccoons and armadillos (Engeman *et al*. 2003), monitor lizards (Tapilatu and Tiwari 2007), mongoose, civets, genets, and ghost crabs (Billes and Fretey 2004), jackals (Hughes 1996), dipteran larvae (Gautreau *et al*. 2008), and army ants (Ikaran *et al*. 2008). In Papua New Guinea, the Huon Coast Leatherback Turtle Conservation Program has successfully reduced dog predation by placing bamboo grids over the nests (Pilcher 2009). Predation on sea turtle hatchlings by birds and fish (see Vose and Shank 2003) has been commonly reported. Reported predation of leatherback hatchlings includes tarpons (Nellis 2000), gray snappers (Vose and Shank 2003), ghost crabs, great blue and yellow-crowned herons, and crested caracaras (Santidrián Tomillo *et al.* 2010). Adult leatherbacks are preyed upon by large predators, such as jaguars, tigers, killer whales, sharks, and crocodiles (reviewed by Eckert *et al.* 2012). Elwen and Leeney (2011) reported a leatherback being ‘tossed and flipped several times by a group of killer whales and then bitten of the flank’ in Walvis Bay, Namibia.

Although disease and predation may pose risk at specific sites, globally they are not known to pose significant risk to leatherback sea turtles.

2.3.2.4 Inadequacy of existing regulatory mechanisms:

The highly migratory nature of leatherbacks requires international collaboration to ensure their survival. Considering their worldwide distribution, virtually every legal instrument that targets or impacts sea turtles is almost certain to cover leatherbacks. A summary of the main global instruments (e.g., regulations, treaties, conventions, agreements) that relate to the conservation and recovery of leatherbacks is provided below.

Since the 2007 5-year review (NMFS and FWS 2007), national regulatory actions include designation of critical habitat off the west coast of the United States in 2012 (see Section 1.3.3). In 2009, the United States established the Mariana Trench, Rose Atoll, and Pacific Remote Islands National Monuments, which prohibited commercial and recreational fisheries in an area encompassing over 95,000 square miles. Under the Magnuson-Stevens Conservation and Management Act (described below), the Hawaii-based shallow-set longline fishery, which targets swordfish, has an annual cap of 26 leatherback interactions each year (50 CFR 665.813). If the limit is reached, the Hawaii-based shallow-set longline fishery is immediately closed and prohibited from shallow-set fishing north of the Equator for the remainder of the calendar year. The fishery was closed in November 2011 after reaching the annual cap.

The conservation and protection of leatherbacks is enhanced by a number of regional and local community conservation programs. Efforts to decrease or eliminate poaching of nesting females and eggs and protect their habitat have been implemented in many areas. For example, the Gabon Sea Turtle Partnership, established in 2005 [http://www.seaturtle.org/groups/gabon/home.html], continues to monitor and protect leatherbacks and their nests. In 2012, the Quarapara Tortugas Marinas Chile was established to promote, coordinate and develop research that contributes to the knowledge and conservation of sea turtles in Chile (Álvarez *et al.* 2013). In the Solomon Islands, landowners displaced by destructive logging practices and concerned about their environment formed a local citizens group to monitor and protect leatherbacks on nesting and foraging grounds (Bero *et al.* 2013). In Papura Barat, Indonesia, efforts to educate local communities about leatherback conservation and provide economic alternatives to enhance conservation are underway (Gjertsen and Pakding 2012). In Papua New Guinea, the Huon Coast Leatherback Turtle Conservation Project utilizes a community development incentive (CDI) strategy through which communities receive financial assistance to levy community committments to promote conservation (Pilcher 2009). Under the CDI, a modest fund is allocated to each village for development purposes to provide tangible benefits to the entire community. These CDIs have included: new furniture for a school, a new church building, materials to complete construction of a church, school or water system, or relief supplies following conflict and strife. The communities decide how to spend the funds in a way which benefits the largest proportion of the village, and the development projects are carried out as collaborative efforts whereby all members of the community and the turtle project come together to fix, repair or build in the name of the community and leatherback conservation. Nonprofit organizations such as the Leatherback Trust [http://www.leatherback.org/about_the_trust.html] have implemented conservation programs and raised awareness on the threats to leatherbacks. These represent only a few examples of regional and local efforts.

As a result of these international, national, and local efforts, many of the anthropogenic threats have been lessened: harvest of eggs and adults has been slowed or virtually eliminated at several nesting areas through nesting beach conservation efforts and an increasing number of community-based initiatives are in place to slow the capture and killing of turtles in foraging areas. Although these efforts need to be maintained to ensure sustainability over time, there is now a more concerted effort to reduce global sea turtle interactions and mortality in artisanal and industrial fishing practices.

United States Magnuson-Stevens Conservation and Management Act

The United States Magnuson-Stevens Fishery Conservation and Management Act (MSA), implemented by NMFS, mandates environmentally responsible fishing practices within federally managed 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."

Bismarck-Soloman Seas Ecoregion: Tri-National Turtle Agreement

In 2006, Indonesia, Papua New Guinea, and the Solomon Islands signed the Tri-National Turtle Agreement to protect leatherback turtles. An action plan was developed and funding was committed to carry forth the conservation program in the region. Additional information is available at: http://www.reffbase.org/pacific/prj_A0000000051.aspx.

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 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 includes 51 European and African States and the European Union as of March 2013. Additional information is available at: http://www.coe.int/t/dg4/cultureheritage/nature/bern/marineturtles/default_en.asp.

Convention on the Conservation of Migratory Species of Wild Animals

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 April 2013, the Convention had 119 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 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 178 Parties as of August 2013. The most recent Parties are the Maldives and Lebanon who signed in 2013. CITES is critically important in ending legal international trade in sea turtle parts. Nevertheless, it does not limit legal and illegal harvest within countries, nor does it regulate intra-country commerce of sea turtle products (Hykle 2002).

The leatherback is listed on Appendices I of CITES as threatened with extinction and international trade is prohibited. Additional information is available at http://www.cites.org.

Convention for the Protection and Development of the Marine Environment of the Wider Caribbean Region

Also called the Cartagena Convention, this instrument has been in place since 1986 and has 23 Signatory States as of March 2013. Under this Convention, the component that may relate to leatherback turtles is the Protocol Concerning Specially Protected Areas and Wildlife (SPAW) that has been in place since 2000. The goals are to encourage Parties "to take all appropriate measures to protect and preserve rare or fragile ecosystems, as well as the habitat of depleted, threatened or endangered species, in the Convention area." 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 the Wider Caribbean Sea Turtle Conservation Network (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 of the North-East Atlantic
The Convention for the Protection of the Marine Environment of the North-East Atlantic (OSPAR) started in 1972 and its purpose is to protect the marine environment of the North-East Atlantic. Fifteen Governments are parties to OSPAR: Belgium, Denmark, Finland, France, Germany, Iceland, Ireland, Luxembourg, The Netherlands, Norway, Portugal, Spain, Sweden, Switzerland and United Kingdom. Under OSPAR Agreement 2008-6, leatherbacks are listed as occurring throughout the OSPAR Convention area (Arctic waters, Greater North Sea, Celtic Seas, Bay of Biscay and Iberian Coast, and Wider Atlantic Ocean) and are under threat and/or in decline in the Convention area. As such, OSPAR is considering adopting a formal recommendation to further the protection and conservation of leatherbacks in the Convention area. Provisions include: (a) review and consider the introduction of national legislation where lacking to protect the leatherback; (b) review existing management programs to determine their effectiveness at protecting leatherbacks and implement further measures where appropriate; (c) promote monitoring and data collection especially fisheries observer programs; (d) raise awareness of the status of and threats to leatherbacks among relevant authorities, fishermen, and general public; and (e) engage the Regional Fisheries Management Organizations to address the threat from high seas fisheries in the International Commission for the Conservation of Atlantic Tunas area.

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 includes 26 Parties as of March 2013. 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.

Food and Agriculture Organization Technical Consultation on Sea Turtle-Fishery Interactions
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 2010 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 InterAmerican Tropical Tuna Commission (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 of a specific mitigation device or bait type in longline fisheries. The InterAmerican Tropical Tuna Commission has also developed a memorandum of understanding with the InterAmerican 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 (http://www.wcpfc.int/node/591).

Other international fisheries organizations that may influence leatherback recovery include the Southeast Atlantic Fisheries Organization (http://www.seafo.org) and the North Atlantic Fisheries Organization (http://www.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. An active sub-regional group for the Western Indian Ocean was created in 2008 under the auspices of the IOSEA and Nairobi Convention, which has improved collaboration amongst sea turtle conservationists in the region (Harris *et al.* 2012). 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.isoeaturtles.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 an 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, 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.

Memorandum of Agreement between Tambrauw-Indonesia and California

On October 15, 2013 a Memorandum of Agreement will be signed by the state of California and Tambrauw-Indonesia (West Papua) to strengthen collaboration and communication regarding the protection of Pacific leatherbacks. The focus will be on outreach, education, funding and support for local conservation efforts in the Monterey Bay National Marine Sanctuary and Abum Marine Protected Area in Bird's Head Seascape (Papua Barat, Indonesia). A 3-day summit is planned for the event and coincides with California's first annual Pacific Leatherback Conservation Day.

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 sea turtles. 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 agreement 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 the 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

This Memorandum of Understanding was concluded under the auspices of the Convention on the Conservation of Migratory Species of Wild Animals and became effective in 1999. It aims at safeguarding six marine turtle species - including the hawksbill - 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. However, despite this agreement, killing adult turtles, harvesting eggs, and turtle bycatch remain widely prevalent along the Atlantic African coast. Additional information is available at http://www.cms.int/species/africa_turtle/AFRICAturtle_bkgd.htm.

Protocol Concerning Specially Protected Areas and Biological Diversity in the Mediterranean

This Protocol is under the auspices of the Barcelona Convention for the Protection of the Mediterranean Sea against Pollution. It 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 this 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.

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 a marine turtle action plan for the Pacific Islands in 2007 and revised the plan in 2012 (http://www.sprep.org).

Tri-Partite Agreement

The Cooperative Agreement for the Conservation of Sea Turtles of the Caribbean Coast of Costa Rica, Nicaragua, and Panama (Tri-Partite Agreement) requires the Parties to work together to protect sea turtle habitats--marine habitats as well as nesting beaches--and to develop and execute a Regional Management Plan to provide guidelines and criteria for a tri-national protected area system for the turtles. Additional information is available at: http://www.conserveturtles.org/velador.php?page=velart13.

Summary

The effectiveness of some of these international instruments varies (Frazier 2008; Hykle 2002; Tiwari 2002). 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, are handicapped by the lack of a sovereign authority to enforce environmental regulations, and/or are not legally-binding. The ineffectiveness of international treaties and national legislation is often times due to the lack of funding, motivation or obligation by countries to implement and enforce them. Further, multi-international agreements often lack ability to share information that would facilitate long-term protection of this transboundary species (Dutton *et al.* 2013b). 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). The legislative framework and management policies of Wider Caribbean countries are comprehensively reviewed by Bräutigam and Eckert (2006).

Notwithstanding the growing number of domestic and intergovernmental authorities, the Services believe that leatherbacks remain in danger of extinction because of the inadequacy of existing regulatory mechanisms for their protection.

2.3.2.5 Other natural or manmade factors affecting its continued existence:

There are also several manmade factors that affect leatherback turtles in foraging areas and on nesting beaches. Two of these are truly global phenomena: climate change and fisheries bycatch. As stated earlier (Section 2.3.2.1), impacts from climate change, especially due to global warming, are likely to become more apparent in future years (IPCC 2007a). The global mean temperature has risen 0.76 ºC over the last 150 years, and the linear trend over the last 50 years is nearly twice that for the last 100 years (IPCC 2007a). There is a high confidence, based on substantial new evidence, that observed changes in marine systems are associated with rising water temperatures, as well as related changes in ice cover, salinity, oxygen levels, and circulation. These changes include shifts in ranges and changes in algal, plankton, and fish abundance (IPCC 2007b).

Climate change will impact sea turtles through increased temperatures, sea-level rise, ocean acidification, changes in precipitation and circulation patterns, and increased cyclonic activity (reviewed by Hamann *et al.* 2013; Poloczanska *et al.* 2009). As global temperatures continue to increase, so will sand temperatures, which in turn will alter the thermal regime of incubating nests and alter natural sex ratios within hatchling cohorts. Because leatherback turtles exhibit temperature-dependent sex determination (reviewed by Wibbels 2003), there may be a skewing of future leatherback cohorts toward a strong female bias since warmer temperatures produce more female embryos (Hawkes *et al*. 2007; Mrosovsky *et al*. 1984). However, because of the tendency of leatherbacks to have individual nest placement preferences and deposit some clutches in the cooler tide zone of beaches, the effects long-term climate change may have on sex ratios may be mitigated (Kamel and Mrosovsky 2004; Patiño-Martinez *et al.* 2012). The effects of global warming are difficult to predict, but changes in reproductive behavior (e.g., remigration intervals, timing and length of nesting season) may occur (reviewed by Hamann *et al.* 2013; Hawkes *et al.* 2009). The pending sea-level rise from global warming is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor. For these areas, the sea or estuarine waters will inundate nesting sites and decrease available nesting habitat (Fish *et al.* 2005; Fonseca *et al.* 2013; Fuentes *et al.* 2010). Sea-level rise is likely to increase the use of shoreline stabilization practices (e.g., sea walls), which may accelerate the loss of suitable nesting habitat. The loss of habitat as a result of climate change could be accelerated due to a combination of other environmental and oceanographic changes such as the frequency and timing of storms and/or changes in prevailing currents, both of which could lead to increased beach loss via erosion (Fuentes and Abbs 2010; Van Houtan and Bass 2007). At sea, hatchling dispersal, adult migration, and prey availability may be affected by changes in surface current and thermohaline circulation patterns (reviewed by Hamann *et al.* 2013; Hawkes *et al.* 2009; Pike 2013). Leatherbacks have extended their range in the Atlantic north by 330km in the last 17 years as warming has caused the northerly migration of the 15°C SST isotherm, the lower limit of thermal tolerance for leatherbacks (McMahon and Hays 2006). Climate change is likely to increase abundance and change the distribution of jellyfish, a major food source for leatherbacks (Attrill *et al.* 2007; Purcell 2005).

Recent studies on the impacts of climate change have focused on specific nesting populations. Saba *et al.* (2012) predict the largest impact from climate change will be on the nesting beaches. They modeled climate conditions at Playa Grande, Costa Rica, and predict the population will remain stable until 2030 and then decline by 75% by 2100. Warmer and dryer conditions will result in lower hatching and emergence success and feminization of eggs. However, these predictions are nullified if a climate-controlled hatchery program is implemented (Saba *et al.* 2012). Santidrián Tomillo *et al.* (2012) used the climate model projections from the IPCC fourth assessment report to estimate the effects of ambient temperature and precipitation on hatch success at Playa Grande, Costa Rica, through 2100. Hatch and emergence success was significantly correlated with weather patterns associated with the ENSO. They estimate that egg and hatchling survival will decline in the region by approximately 50-60% by the end of this century (Santidrián Tomillo *et al.* 2012). Fuentes *et al.* (2013) surveyed sea turtle experts to determine the likely resilience regional populations of sea turtles would have to climate change. They found that the experts felt that the southwest Atlantic and Indian Ocean leatherback populations were the least resilient to climate change. Fisheries bycatch may be a major factor in the ability for these stocks to withstand the added stress of climate change. In his climate-based Population Viability Analysis model, Van Houtan (2011) concluded that the decline in the Jamursba-Medi nesting population since 1993 is likely linked to climate. Ussa (2013) predicted a 20-25% loss in beach areas due to sea level rise by the year 2100 within the Archie Carr National Wildlife Refuge, Florida, as well as adjacent areas that are being considered for acquisition into the Refuge.

A significant factor impacting leatherback populations worldwide is incidental capture in artisanal and commercial fisheries (reviewed by Eckert *et al.* 2012; Lewison *et al*. 2004, 2013; Wallace *et al.* 2010a, 2013). Globally, over 85,000 sea turtles (all species combined) are estimated to be bycaught in fisheries deploying gill nets, longlines and trawls (Wallace *et al.* 2010a). Pelagic longlines were estimated to take more than 50,000 leatherbacks worldwide in 2000 (Lewison *et al*. 2004). Small-scale coastal fisheries are a major component of the global bycatch. Of the estimated 51 million people employed in fisheries worldwide, over 99% operate in non-industrial coastal fisheries (Peckham *et al.* 2007). Small-scale fisheries are reported to have significant ecological impacts due to their high bycatch discards and benthic habitat destruction (Shester and Micheli 2011). To date, the highest sea turtle bycatch rates and levels of observed effort exist in the East Pacific, Northwest and Southwest Atlantic, and Mediterranean regions, but there also exists significant data gaps around Africa, in the Indian Ocean, and Southeast Asia where high bycatch rates have also been documented in coastal trawl, net and longline fisheries (Wallace *et al.* 2013). Coastal artisanal fisheries are a major concern for bycatch of sea turtles as well as ecological impacts to the marine environment.

In the United States, Finkbeiner *et al.* (2011) analyzed incidental take across all commercial fisheries from 1990 through 2007. They examined take reduction based on the year a particular fishery implemented bycatch reduction measures. Prior to implementing bycatch reduction measures, approximately 3,900 live and 2,350 dead leatherbacks were taken in U.S. commercial fisheries each year. After implementing reduction measures, 1,430 live and 50 dead leatherbacks were estimated to be taken annually (Finkbeiner *et al.* 2011). Since 2005, U.S. longliners have been required to use safe-handling techniques in an effort to reduce mortality and increase survivorship of incidentally captured sea turtles. The U.S. Hawaii-based shallow-set longline fishery has dramatically reduced sea turtle interactions since requiring large circle hooks and non-squid bait, employing annual interaction limits resulting in fishery closures if limits are reached, and hosting protected species workshops to raise awareness and provide education to vessel operators and owners (NMFS 2012a). Additionally, NMFS has supported numerous international programs or provided in-country training to build capacity for observer programs and to promote longline bycatch reduction gear technology and safe turtle-handling practices. Five International Fisheries Forums have also been held to promote the transfer and uptake of bycatch reduction technology to international longline fleets of the Pacific. Since 2001, a large time/area closure was put in place off the U.S. west coast to reduce leatherback interactions with the California large mesh drift gillnet fishery. The time/area closure has resulted in a significant reduction in leatherback entanglements. Larger turtle excluder devices were required in 2003 in shrimp trawl gear in areas of the U.S. to allow larger turtles, including leatherbacks, to pass through (Epperly and Teas 2002; NMFS 2002). However, significant interactions with the shrimp trawl fleet occur. An estimated 1,628 interactions/captures and 144 leatherback deaths occur each year in the shrimp fishery (otter trawl, skimmer trawl, and trynets combined; NMFS 2012b, Table 33). In the northeast United States, 110 live and 27 dead leatherbacks were reported entangled in fishing gear (e.g., weirs, lobster pots) between 2007 and 2012 (Sea Turtle Disentanglement Netowrk, unpublished data).

Lee Lum (2006) estimated that more than 3,000 leatherbacks were entangled by coastal gillnets off Trinidad in the Southern Caribbean annually, with a 30% mortality. Despite numerous Argentinian regulations and international instruments to protect sea turtles, significant bycatch still occurs in artisanal and commercial fisheries operating in territorial waters of Argentina, Uruguay, and Brazil and the highseas (González-Carmen *et al.* 2012). Riskas and Tiwari (2013) characterized predominant fisheries and sea turtle bycatch for 21 nations along the Atlantic coast of Africa from Mauritania south to Namibia. Leatherback incidental take was reported for 17 nations indicating extensive and high bycatch in the region. Longline and driftnet fisheries in Moroccan waters off northwest Africa are reported to capture approximately 100 leatherbacks per year (Benhardouze *et al.* (2012). Nel *et al.* (2013) cite commercial and artisanal fisheries operating off South Africa as a major concern for the leatherback nesting population in South Africa. The decline in the Mexican population of leatherbacks has been suggested to coincide with the growth of the longline and coastal gillnet fisheries in the Pacific (Eckert and Sarti 1997); leatherbacks from this population migrate to the north Pacific and southeastern Pacific where these fisheries operate (Dutton *et al*. 2000b; Eckert and Sarti 1997; Sarti Martinez *et al*. 2007). High mortality is documented for leatherbacks caught in artisanal nearshore fisheries in Peru (Alfaro-Shigueto *et al.* 2007, 2011; Paredes and Quiñones 2013). Between 2001 and 2005, the Chilean swordfish-directed fishery was observed to take 284 leatherbacks on 7,976 sets (CPUE was low 0.0268 when compared to other studies) (Donoso and Dutton 2010). However, when combined with other fisheries and threats in this region, the Chilean swordfish fishery remains a concern for the severely depleted nesting populations in the eastern Pacific Ocean (Donoso and Dutton 2010).

The need to reduce bycatch of leatherbacks has led to many experiments and new insights. For example, Senko *et al.* (2013) examined fisheries bycatch mitigation measures (time-area closures, bycatch limits, gear modifications, and buy-outs) for their efficiency in reducing leatherback bycatch. They determined that time-area closures were the least effective because closures either did not encompass the entire geographic area needed for conservation or re-distributed the bycatch to other fisheries outside of the management area. Gear modifications were the most effective at reducing bycatch, at least in cases where a single fishery was the source of the high bycatch. Gear modifications are less likely to re-distribute the bycatch and are more likely to result in greater industry buy-in either through engaging fishers in developing and testing the gear modifications or by allowing fishers to fish in areas that would otherwise be closed (Senko *et al.* 2013). For instance, in the pelagic longline fishery circle hooks with squid bait and J and circle hooks with mackerel bait were found to greatly reduce leatherback bycatch (Watson *et al*. 2005). Fewer leatherbacks were caught on large circle hooks when compared to J-hooks used in pelagic longline sets targeting swordfish and bigeye tuna in the Atlantic Ocean (Pacheco *et al.* 2011). Andraka *et al.* (2013) examined the performance of circle hooks in relation to J-style and tuna hooks on target catch and sea turtle bycatch in fisheries from Ecuador, Panama, and Costa Rica and found that circle hooks reduced sea turtle bycatch. Gless and Salmon (2008) evaluated the responses of juvenile leatherbacks to lights used in longlines and found behaviorally complex reactions as they showed elements of attraction and repulsion to this stimulus. Furthermore, the potential post-hooking mortality can be significantly reduced with tools to remove hooks and line from the turtles (Watson *et al*. 2005).

Among the anthropogenic factors affecting leatherbacks, boat strikes (Dwyer *et al*. 2003; Foley *et al.* 2009; Turtle Expert Working Group 2007) and the ingestion of plastics, balloons, synthetic materials, and fishing hooks and nets have been reported (Barreiros and Barcelos 2001; Bugoni *et al*. 2001; Duguy *et al*. 1998; Plot and Georges 2010; Poppi *et al.* 2012; Starbird and Audel 2000). Autopsies of leatherbacks dating back to 1885 through 2007 (n = 408), showed a marked increase in the presence of plastic in the intestinal tract beginning in 1968, when the first record of plastic ingestion was reported (Mrosovsky *et al.* 2009). Plastics were present in over 30% (n = 138) of the autopsy reports and, of these reports, approximately 9% had plastic in amounts and location that appeared to obstruct the passage of food and likely was the cause of death (Mrosovsky *et al.* 2009). Ingestion of marine debris can result in starvation, gut strangulation, and toxicity; however, more studies are needed to determine the physiological effects of ingesting these materials.

Increased exposure to heavy metals and other contaminants in the marine environment also affect leatherbacks, albeit perhaps not as globally significant as those mentioned above. Organochlorine contaminants, perfluoroalkyl compounds, cadmium, copper, zinc, and toxic metals have been identified in leatherbacks, but it is difficult to interpret their effect on the health of this endangered species (Caurant *et al*. 1999; Godley *et al*. 1998; Keller *et al.* 2012; McKenzie *et al*. 1999; Orós *et al.* 2009; Poppi *et al.* 2012; Storelli and Marcotrigiano 2003). Guirlet (2005) found high levels of organochloride pesticides in the sand of a French Guiana nesting beach, which may explain low hatching success on this beach (Girondot *et al*. 2007). Keller (2013) reviewed the studies on persistent organic pollutants (i.e., is carbon-based and persist for long periods in the environment) and clearly demonstrated that sea turtles are exposed to these pollutants depending on the species and location. Across all studies and species, classes of polychlorinated biphenyls had the highest concentrations and classes of hexachlorobenzene and hexachlorohexanes had the lowest concentrations in samples taken from sea turtles (reviewed by Keller 2013).

Contaminants have been found to pass from nesting females to their eggs, which partially may explain poor hatching and emergence success, a characteristic of the species (reviewed by Eckert *et al.* 2012; Guirlet *et al.* 2008, 2010; Perrault *et al.* 2011; Stewart *et al*. 2008, 2011b). Nesting females transferred selenium and mercury to their offspring in nests laid in Florida (Perrault *et al.* 2011). Hatchlings were found to have heart and skeletal degeneration indicative of selenium-deficient mothers. Selenium deficiency can result from ingestion of high levels of mercury, which is detoxified through the liver by formation of a mercury-selenium compound. Exposure to mercury, over time, decreases the liver's ability to detoxify the mercury. Perrault *et al.* (2011) found that hatching and emergence success was greater for hatchlings with elevated liver selenium and mercury-selenium compounds. Mercury and selenium concentrations increase in leatherbacks as they age (Perrault 2013). Mercury and selenium concentrations in the blood vary between females nesting in Florida and those nesting at Sandy Point National Wildlife Refuge. These differences may be attributed to divergent migratory routes to foraging grounds (Perrault *et al.* 2011, 2013).

Based on the information described above, the Services conclude that leatherbacks remain in danger of extinction because of other natural or manmade factors affecting their continued existence.

2.4 Synthesis

Recent research has added to our knowledge of how leatherback sea turtles interact with their environment and how they contribute to a healthy marine ecosystem. We know more now about their migration patterns. We have a better understanding of the biological and environmental factors that influence where leatherbacks forage. The results of long-term studies have filled gaps in our understanding of demography and population structure. Advances in genetic and stable isotope analyses, tagging techniques, especially satellite, radio, and sonic telemetry have vastly improved our knowledge of the biology and ecology of leatherback sea turtles. Understanding the ecological role leatherbacks hold in their environment and predicting where they are in space and time are important for developing management strategies to meet recovery goals and objectives (see section 2.3.1).

The East Pacific and Malaysia leatherback populations have collapsed, yet Atlantic populations generally appear to be stable or increasing. Many explanations have been provided to explain the disparate population trends, including fecundity and foraging differences seen in the Pacific, Atlantic, and Indian Oceans. Since the last 5-year review, studies indicate that high reproductive output and consistent and high quality foraging areas in the Atlantic Ocean have contributed to the stable or recovering populations; whereas prey abundance and distribution may be more patchy in the Pacific Ocean, making it difficult for leatherbacks to meet their energetic demands and lowering their reproductive output (see section 2.3.1.2).

Both natural and anthropogenic threats to nesting and marine habitats continue to affect leatherback populations, including the 2004 tsunami in the Indian Ocean, 2010 oil spill in the U.S. Gulf of Mexico, logging practices, development, and tourism impacts on nesting beaches in several countries. Egg collection continues to occur in many countries around the world and has been attributed to catastrophic declines in some areas. Although greatly reduced, the killing of nesting females still remains a matter of concern on many beaches. A wide variety of species depredate leatherback nests worldwide (e.g., feral pigs and dogs). Climate change is an emerging and major threat to the conservation and recovery of leatherbacks. The sea level is expected to rise resulting in the loss of nesting habitat. The ocean is expected to become warmer impacting ocean processes and prey abundance and distribution. Average air temperatures are expected to be warmer, thus exposing leatherback eggs to hotter temperatures and skewing natural sex ratios. Loss of suitable nesting habitat is anticipated to continue as coastal areas are developed. Although gear modification and fisheries practices implemented in many fisheries have reduced incidental bycatch of leatherbacks, globally artisanal and commercial fishing operations are major impacts that are far from being resolved. Additional factors affecting leatherbacks include boat strikes, the ingestion of and entanglement in marine debris, and exposure to heavy metals and other contaminants in the nesting and marine environments (see sections 2.3.2.1 through 2.3.2.3 and 2.3.2.5).

Several international agreements provide legal protection for sea turtles; however, additional multi-lateral efforts are needed to ensure they are sufficiently implemented and/or strengthened, and key non-signatory parties need to be encouraged to accede. The effectiveness of some of these international instruments varies due to many factors such as participation, funding, and compliance (see section 2.3.2.4).

3.0 RESULTS

3.1 Recommended Classification:

Based on the best available information, we do not believe the leatherback turtle should be delisted or reclassified. However, we have information that indicates an analysis and review of the species should be conducted in the future to determine the application of the DPS policy to the leatherback turtle. See Section 4.0 for additional information.

3.2 New Recovery Priority Number: No change.

4.0 RECOMMENDATIONS FOR FUTURE ACTIONS

We have preliminary information that indicates an analysis and review of the species should be conducted in the future to determine the application of the DPS policy to the leatherback. Since the species' listing, a substantial amount of information has become available on population structure (through genetic studies) and distribution (through telemetry, tagging, stable isotope, and genetic studies). The Services have not yet fully assembled or analyzed this new information; however, at a minimum, these data appear to indicate a possible separation of populations by ocean basins. To determine the application of the DPS policy to the leatherback, the Services intend to fully assemble and analyze this new information in accordance with the DPS policy. See Section 2.3 for new information since the last 5-year review.

The Services recommend the recovery plans be re-examined over the next 5-year horizon, particularly if the DPS analysis results in restructuring of the current listing, to update the plans to conform to current recovery planning guidance. The current "Recovery Plan for Leatherback Turtles (*Dermochelys coriacea*) in the U.S. Caribbean, Atlantic, and Gulf of Mexico" was signed in 1992 and the "Recovery Plan for U.S. Pacific Populations of the Leatherback Turtle (*Dermochelys coriacea*)" was signed in 1998. The recovery plans are dated and do not address a major, emerging threat—climate change. Actions to protect nesting beaches and foraging habitat and to preserve natural sex ratios should be understood in terms of impacts from climate change. Those plans should conform to the Services' Interim Recovery Planning Guidance (http://www.nmfs.noaa.gov/pr/pdfs/recovery/guidance.pdf) and comprehensively examine the threat of climate change and develop local actions, if possible, to minimize the impacts.

The Services recommend that research continue and be made a priority, which provides information on long-term population trends based on both nesting and in-water population monitoring (National Research Council 2010), hatchling and juvenile dispersal, genetic relationships among nesting populations, impacts of and bycatch reduction from coastal and pelagic fisheries, impacts of climate change, and identification of and threats at foraging areas.

The Services recommend that federal grant programs, relevant to sea turtle conservation and protection, continue to support efforts in the Atlantic Ocean and prioritize support for conservation and protection programs that would most benefit leatherback populations in the Pacific Ocean.

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NATIONAL MARINE FISHERIES SERVICE
5-YEAR REVIEW of *Leatherback Sea Turtle*

Current Classification: Endangered

Recommendation resulting from the 5-Year Review: No change

Review Conducted By:

Therese Conant, Angela Somma (National Marine Fisheries Service)
Ann Marie Lauritsen, Kelly Bibb (U.S. Fish and Wildlife Service)

REGIONAL OFFICE APPROVAL: The draft document was reviewed by the appropriate Regional Offices and Science Centers.

HEADQUARTERS APPROVAL:
Director, Office of Protected Resources, NOAA Fisheries

Approve: Donna S Wieting Date: OCT 30 2013
Donna S. Wieting

Assistant Administrator, NOAA Fisheries

_____ Concur _____ Do Not Concur

Signature________________ Date 11/15/13

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Deputy Assistant Administrator
for Regulatory Programs,
performing the functions and duties of the
Assistant Administrator for Fisheries

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FIELD OFFICE APPROVAL:

Lead Field Supervisor, Fish and Wildlife Service

Approve ____________________ Date 11/13/13

REGIONAL OFFICE APPROVAL:

for
Lead Regional Director, Fish and Wildlife Service

Approve ____________________ Date 11-11-13