OLIVE RIDLEY SEA TURTLE
(*LEPIDOCHELYS OLIVACEA*)

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 FIELD OFFICE
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U.S. FISH AND WILDLIFE SERVICE

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1.0 GENERAL INFORMATION

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1.2. Methodology Used to Complete the Review

A 5-year review is a periodic analysis of a species' status conducted to ensure that the listing classification of a species as threatened or endangered on the List of Endangered and Threatened Wildlife and Plants (List) (50 CFR 17.11 – 17.12) is accurate. The 5-year review is required by section 4(c)(2) of the Endangered Species Act of 1973, as amended (ESA). To achieve this, 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. Information sources include the final rule listing this species under the Endangered Species Act (ESA); the recovery plan for the U.S. Pacific populations; peer reviewed publications; unpublished field observations by the Services, States, and other experienced biologists; unpublished survey reports; and notes and communications from other qualified biologists.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, 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. The information on the olive ridley biology and habitat, threats, and conservation efforts were summarized and analyzed in light of the recovery criteria and the ESA section 4(a)(1) factors (see Sections A.2.3.2.1 and B.2.3.2.1) to determine whether a reclassification or delisting is warranted (see Section 3.0). If the recovery criteria do not meet the 2006 NMFS Interim Recovery Planning Guidance or do not adequately address new threats, then we use the criteria only as a benchmark for measuring progress toward recovery. In the case where there is no recovery plan (i.e., Atlantic Ocean), we rely on the ESA section 4(a)(1) factors analysis.

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

Original Listing
FR notice: 43 FR 32800
Date listed: July 28, 1978
Classification and Entity listed: 2 populations or groups of populations

> *Endangered Populations* – breeding colony populations on Pacific coast of Mexico
> *Threatened Populations* – wherever found except where listed as Endangered

1.3.3 Associated rulemakings

There are no associated rulemakings with the original listing.

1.3.4 Review history

National Marine Fisheries Service and U.S. Fish and Wildlife Service. 2007. Olive Ridley Sea Turtle (*Lepidochelys olivacea*) 5-Year Review: Summary and Evaluation. National Marine Fisheries Service, Silver Spring, Maryland. 64 pages.

> Conclusion: Retain the listing as Endangered for the breeding colony populations on Pacific coast of Mexico and the Threatened listing wherever found except where listed as endangered. 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 Endangered for the breeding colony populations on Pacific coast of Mexico and the Threatened listing wherever found except where listed as endangered. [*Note*: the status review concluded that the olive ridley in the western Atlantic should be listed as Endangered. However, populations in the western Atlantic were not listed separately from the global listing. A Distinct Population Segment analysis for the western Atlantic was not conducted as a result of the 1995 status review.]

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 Endangered for the breeding colony populations on Pacific coast of Mexico and the Threatened listing wherever found except where listed as endangered. [*Note*: the status review concluded that the olive ridley in the western Atlantic (i.e., Suriname and adjacent areas) should be listed as Endangered. However,

populations in the western Atlantic were not listed separately from the global listing. A Distinct Population Segment analysis for the western Atlantic was not conducted as a result of the 1985 status review.]

FWS also conducted 5-year reviews for the olive ridley in 1983 (48 FR 55100) 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.

Conclusion: Retain the listing as Endangered for the breeding colony populations on Pacific coast of Mexico and the Threatened listing wherever found except where listed as endangered.

1.3.5 Species' recovery priority number at start of review

National Marine Fisheries Service = 5 (this represents a moderate magnitude of threat, a high recovery potential, and the presence of conflict with economic activities).

U.S. Fish and Wildlife Service (48 FR 43098) = 8C (this represents a full species with a moderate degree of threat, a high recovery potential, and the potential for conflict with construction or other development projects or other forms of economic activity).

1.3.6 Recovery plan

Name of plan: Recovery Plan for U.S. Pacific Populations of the Olive Ridley Turtle (*Lepidochelys olivacea*) (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, it is listed as two populations. Those populations were listed in 1978 before the ability to list a DPS was added to ESA.

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, we noted information indicating an analysis and review of the species should be conducted in the future to determine the application of the DPS Policy to the olive ridley. 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 "Recovery Plan for U.S. Pacific Populations of the Olive Ridley Turtle (*Lepidochelys olivacea*)" was signed in 1998, and while not all of the recovery criteria strictly adhere to all elements of the NMFS Interim Recovery Planning Guidance, they provide a useful benchmark for measuring progress toward recovery. Thus, we consider progress toward recovery objectives in this section. Also, the 1998 recovery plan does not explicitly identify downlisting criteria for the Endangered breeding colony populations on the Pacific coast of Mexico. Rather, the plan states that the primary threat (i.e., harvest) to the breeding colony populations on the Pacific coast of Mexico had been reduced and downlisting to Threatened may be feasible. Finally, there is no recovery plan for the olive ridley in the Atlantic Ocean as their occurrence is largely outside of U.S. jurisdiction.

Recovery Objectives as written in the Recovery Plan for U.S. Pacific Populations of the Olive Ridley Turtle

The recovery criteria are identified below, along with several key accomplishments:

To consider delisting, all of the following criteria must be met:

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

    Status: This criterion is partially complete. Stock structure of nesting turtles in the Pacific Ocean has been identified using genetic analysis, flipper tagging, and satellite telemetry. 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.

2. Foraging populations are statistically significantly increasing at several key foraging grounds within each stock region.

Status: Efforts to attain this criterion are ongoing. Population abundance has been surveyed and data on size, diet, and distribution of olive ridleys has been collected in the eastern tropical Pacific during NOAA research cruises. The Stable Isotope Laboratory at NMFS Southwest Fisheries Science Center maintains over 1,500 sea turtle tissue and environmental (i.e., dietary) samples, the vast majority of which have been analyzed for stable-carbon and -nitrogen isotopes. Stable isotope analysis will help identify foraging grounds and facilitate monitoring population trends on those grounds.

3. All females estimated to nest annually at "source beaches" are either stable or increasing for over 10 years.

   Status: Efforts to attain this criterion are ongoing. Based on the number of olive ridleys nesting on the Pacific coast of Mexico, the Endangered population appears to be stable at some locations (e.g., Mismaloya, Tlacoyunque, and Moro Ayuta), increasing at La Escobilla and Ixtapilla, and decreasing at Chacahua. A comparison of the current abundance of the Mexico nesting assemblages with the former abundance at each of the large arribada beaches indicates that the populations experienced steep declines that have not yet been overcome. Nesting trends in Mexico at several non-arribada beaches are stable or increasing since 1999.

4. A management plan based on maintaining sustained populations for turtles is in effect.

   Status: Not yet completed.

5. International agreements are in place to protect shared stocks.

   Status: This criterion is partially complete. The U.S. is a party to the Inter-American Convention for the Protection and Conservation of Sea Turtles. However, incidental capture in high-seas fisheries remains a concern. See Sections A.2.3.2.4 and B.2.3.2.4 for discussion of international instruments in place to protect sea turtle populations.

2.3 Updated Information and Current Species Status

The review is based on new information since the 2007 review and through January 2014. The review does not generate new data through research or modeling and is not an exhaustive review of what is known about the olive ridley sea turtle. Rather, it provides an overview of the information on olive ridley biology, population distribution and trends, habitat, and threats that have emerged since the last 5-year review (NMFS and FWS 2007). The objective of the review is to assess whether the current listing classifications for the olive ridley sea turtle is appropriate. The section is divided into two subsections (A and B) based on the current listings -- Subsection A refers to the 'Endangered' breeding colony populations on the Pacific coast of Mexico and Subsection B refers to the 'Threatened' populations. As such, there is some repetition of information in the two subsections because the information either pertains to both listings or the origin of the affected individual turtles cannot be identified to a listed population due to in-water mixing of populations.

SUBSECTION A: ENDANGERED POPULATIONS

A.2.3.1 Biology and Habitat

A.2.3.1.1. Distribution

The olive ridley has a circumtropical distribution, occurring in the Atlantic, Pacific, and Indian Oceans (Pritchard 1969). For distribution maps see State of the Worlds Sea Turtles OBIS-SEAMAP: http://seaturtlestatus.org. Globally, olive ridleys are found in coastal waters of over 80 countries (Abreu-Grobois and Plotkin 2008). They do not nest in the United States. In the eastern Pacific, olive ridleys typically occur in tropical and subtropical waters, as far south as Peru and as far north as California, but occasionally have been documented as far north as Alaska (Hodge and Wing 2000). The Endangered breeding colony populations on the Pacific coast of Mexico include key arribada nesting beaches at Mismaloya, Ixtapilla, and La Escobilla. Solitary nesting occurs along the entire Pacific Ocean coast of Mexico. See Section A.2.3.1.6 Abundance and Population Trends (Table 1) for further details.

A.2.3.1.2. Migration

Within a region, olive ridleys may move between the oceanic zone and the neritic zone (Plotkin *et al*. 1995; Shanker *et al*. 2003a) or only occupy neritic waters (Pritchard 1976; Reichart 1993). Olive ridleys are not known to move between or among ocean basins. Thus, for the purposes of the Endangered populations section, we examine movement only within the central and eastern Pacific Ocean assuming that the majority of information in the western Pacific Ocean and southeast Asia are from turtles that originate from breeding colonies listed as Threatened (see Section B.2.3.1). However, this section also covers Threatened populations in Costa Rica, Panama, and elsewhere in the eastern Pacific because the breeding origin of olive ridleys is not always known.

In the eastern Pacific, olive ridleys are highly migratory and appear to spend most of their non-breeding life cycle in the oceanic zone (Arenas and Hall 1992; Beavers and Cassano 1996; Cornelius and Robinson 1986; Pitman 1991, 1993; Plotkin 1994, 2010; Plotkin *et al*. 1994, 1995). Olive ridleys occupy the neritic zone during the breeding season. Some reproductively active males and females migrate toward the coast and aggregate at nearshore breeding grounds located near nesting beaches (Cornelius 1986; Hughes and Richard 1974; Kalb *et al*. 1995; Pritchard 1969; Plotkin *et al*. 1991, 1996, 1997). A significant proportion of the breeding also takes place far from shore (Kopitsky *et al*. 2000; Pitman 1991), and some males and females may not migrate to nearshore breeding aggregations at all. Some males appear to remain in oceanic waters, are non-aggregated, and mate opportunistically as they intercept females *en route* to near shore breeding grounds and nesting beaches (Kopitsky *et al*. 2000; Parker *et al.* 2003; Plotkin 1994; Plotkin, *et al*. 1994, 1996).

The post-reproductive migrations of olive ridleys in the eastern Pacific Ocean are unique and complex. Their migratory pathways vary annually (Plotkin 1994, 2010), there is no spatial and temporal overlap in migratory pathways among groups or cohorts of turtles (Plotkin *et al*. 1994, 1995), and no apparent migration corridors exist (Plotkin 2010). Unlike other sea turtles that migrate from a breeding ground to a single feeding area, where they reside until the next breeding season, olive ridleys are nomadic migrants that swim hundreds to thousands of kilometers over vast oceanic areas (Parker *et al.* 2003; Plotkin 1994, 2010; Plotkin *et al*. 1994, 1995). This nomadic behavior may be unique to olive ridleys in the eastern Pacific Ocean as studies in other ocean basins indicate olive ridleys occupy neritic waters and do not make the extensive migrations observed in this region (Plotkin 2010).

Polovina *et al.* (2003, 2004) tracked 10 olive ridleys caught in the Hawaii-based pelagic longline fishery. The olive ridleys identified as originating from the eastern Pacific populations stayed south of major currents in the central North Pacific-southern edge of the Kuroshio Extension Current, North Equatorial Current, and Equatorial Counter Current; whereas, olive ridleys identified from the western Pacific associated with these major currents, suggesting that olive ridleys from different populations may occupy different oceanic habitats (Polovina *et al.* 2003, 2004).

Data are lacking on post-hatchling and juvenile dispersal. Pitman (1990) observed sea turtles from vessels from 1975 through 1990 for a total of 4,179 days at-sea. He found the olive ridley to be the most common turtle south of the Baja Peninsula, and many sightings were of adults mating. It is unknown what portion of sightings were juveniles, if any. Eleven juveniles were sighted in the Revillagigedo Archipelago offshore Mexico during four surveys carried out between November 1999 and December 2000. All were sighted in deep, pelagic water and algae had not accumulated on their carapaces, indicating offshore habitat use (Juárez-Cerón and Sarti-Martínez, 2003).

A.2.3.1.3. Demography

Data are lacking on the demography of this species. Several aspects of demography likely are shared between the Endangered and Threatened populations. Given the paucity of information and the possible shared characteristics, this section examines what is known on a species level.

Survival

Other than density-dependent survivorship of olive ridley eggs and emergent hatchlings (see Reproductive Capacity below), data are lacking on post-hatchling and other life stage survival rates (Abreu-Grobois and Plotkin 2008). Presumably, similar to other sea turtles, olive ridleys experience high mortality in their early life stages (Abreu-Grobois and Plotkin 2008).

Growth and Age at Maturity

Growth rate data for olive ridleys in the wild are unknown (Avens and Snover 2013; NMFS and FWS 1998). Female olive ridleys are believed to attain sexual maturity at an age similar to its congener, the Kemp's ridley (*Lepidochelys kempii*). Based on samples collected in the north-central Pacific Ocean, Zug *et al*. (2006) estimated the median age of sexual maturity for the olive ridley is 13 years with a range of 10 to 18 years.

Reproductive Capacity

Individual olive ridleys exhibit three different reproductive behaviors: mass or arribada nesting, dispersed or solitary nesting, and a mixed strategy of both (Bernardo and Plotkin 2007; Fonseca *et al.* 2013; Kalb 1999). Olive ridleys commonly nest in successive years (Cornelius 1986; Plotkin 1994; Pritchard 1969), and the behavior may well be the norm for the species. In general, individual olive ridleys may nest one, two, or three times per season but on average two clutches are produced annually, with approximately 100-110 eggs per clutch (Pritchard and Plotkin 1995). However, smaller females may produce fewer eggs per clutch (Harfush *et al.* 2008a).

Reproductive characteristics may differ between arribada and solitary nesters. Multiple paternity (i.e., more than one male fertilizing eggs in a clutch) was significantly greater in nests from arribada beaches, which may be attributed to population size and the associated increase in male encounter rates (Jensen *et al*. 2006). At Nancite Beach, Costa Rica, arribada nesters produced significantly larger clutches (i.e., more eggs) compared to solitary nesters, although other characteristics such as female size, egg size, or within-clutch variability in egg size, were not different between the groups (Plotkin and Bernardo 2003). Smaller clutch sizes observed for solitary nesters might be due to energetic costs associated with undertaking internesting movements among multiple beaches (Plotkin and Bernardo 2003). Solitary nesters generally oviposit on 14-day cycles whereas arribada nesters oviposit approximately every 28 days (Kalb 1999; Kalb and Owens 1994; Pritchard 1969). However, this generality may not apply to all populations. Solitary nesters in Sergipe, Brazil, averaged 22.35 $\pm$ 7.01-days internesting cycle (Matos *et al.* 2012). Within a nesting season, solitary nesters use multiple beaches for oviposition but arribada nesters display nest site fidelity (Kalb 1999). However, several studies indicate this, too, may not apply to all populations--some arribada nesters nest at different arribada beaches (Fonseca *et al.* 2013; Shanker *et al*. 2003b), and some solitary nesters show strong site fidelity (Whiting *et al.* 2007a).

Nest success varies in time and space. On solitary nesting beaches, where density-dependent mortality is not a factor, hatching rates are significantly higher (Castro 1986; Dornfeld and Paladino 2012; Gaos *et al*. 2006). Conversely, survivorship is low on high density arribada nesting beaches because of density-dependent mortality (Cornelius *et al*. 1991). The sheer number of turtles (1,000-500,000 turtles) nesting in spatially limited areas results in density-dependent egg mortality during a single arribada. Moreover, turtles return approximately every month during a discrete nesting season (3-6 months) and nests that remained intact during the previous month are again at risk when new waves of turtles crawl ashore. For example, at La Escobilla, Mexico, approximately 6% of nests were destroyed in the first arribada, but increased to over 15% in the second arribada as nest density increased (Ocana *et al.* 2012). In addition to nest disturbance, the existence of high nest densities over time apparently alters the nutrient composition of sand, as well as the concentration of ammonia in the sand (McPherson and Kibler 2008). High ammonia concentrations, and/or high concentrations of fungal and bacterial pathogens, at beaches with high nest densities might also contribute to density-dependent nest loss. In controlled experiments at Playa La Flor, Nicaragua, and Playa Nancite and Ostional, Costa Rica, nest density affected hatching success with higher density resulting in lower hatching success (Bézy *et al.* 2013; Honarvar 2007; Honarvar *et al.* 2008). As nest density increased, gas exchange became limited during the latter part of the incubation period, likely due to the increased metabolic activity from developing embryos. CO_2 levels increased and O_2 levels decreased in higher density plots, which led to higher embryo death (Honarvar 2007; Honarvar *et al.* 2008). Bacterial (Honarvar *et al.* 2011) and small organism (Madden *et al*. 2008) diversity and richness were also greater in areas of high nest density and close to vegetation and away from tidal wash. During high-density arribadas, nesting females inadvertently break eggs, which provide nutrients for increased bacterial growth. Also, the high zone on the beach is less likely to be exposed to tidal overwash and accumulation of broken eggs in this area over time may contribute to bacterial diversity and richness (Honarvar *et al.* 2011).

Sex Ratios

Olive ridleys exhibit temperature-dependent sex determination, and warmer incubation temperatures produce more females (reviewed by Wibbels 2003, 2007). The middle third of the incubation period is when the developing embryo's sex determination is sensitive to temperatures (Merchant-Larios *et al.* 1997). The temperature at which a nest will produce 50% males/females was estimated to be 29.95°C for nesting populations in Mexico (Sandoval-Espinoza 2011 as cited in Hernández-Echeagaray *et al.* 2012), approximately 30-31°C for nesting populations in Costa Rica, and less than 29°C in Gahirmatha, India (reviewed by Wibbels 2007). Pivotal temperatures likely vary within and among populations and generalizations should be applied with caution.

Studies on sex ratios of olive ridley hatchlings are few and non-existent for juvenile and adults. Hernández -Echeagaray *et al*. (2012) found a slight female-bias sex ratio (55%) for the 2010-2011 nesting season at La Escobilla, Mexico. Sex ratios may also change over the nesting season. In Mexico, a female-biased hatchling sex ratio was found at most nest sites (La Escobilla was not included) at the beginning of the nesting season, and a male-biased ratio at the end of the season (Sandoval-Espinoza 2011 as cited in Hernández-Echeagaray *et al.* 2012).

A.2.3.1.4. Taxonomy, Phylogeny, and Genetics

For the purposes of this section, we include both the Endangered and Threatened populations as they are not species, or sub-species, and were listed prior to the the 1978 amendment to the ESA, which included distinct population segment in the definition of species. The olive ridley taxonomic classification (below) is unchanged since the last 5-year review (NMFS and FWS 2007).

Kingdom:	Animalia
Phylum:	Chordata
Class:	Reptilia
Order:	Testudines
Family:	Cheloniidae
Genus:	*Lepidochelys*
Species:	*olivacea*
Common names:	Olive ridley sea turtle, Pacific ridley sea turtle

Intra-specific phylogeographic differentiation occurs among, as well as within, ocean basins (Bowen *et al*. 1998; Hahn *et al.* 2012; Lopez-Castro and Rocha-Olivares 2005; Shanker *et al*. 2004). Four main lineages are identified: east India (believed to be the ancestral lineage), the Indo-Western Pacific lineage, the Atlantic lineage, and the eastern Pacific lineage (Bowen *et al*. 1998; Hahn *et al.* 2012; Shanker *et al*. 2004). The ancestral split between the Indian Ocean and other basins likely occurred approximately 2.7 million years ago (Duchene *et al.* 2012).

Within these lineages, few in-depth genetic surveys have assessed fine-scale population structure. Several studies found moderate to high genetic differentiation among regional rookeries separated by more than 500 km, but low differentiation for rookeries in closer proximity (e.g., Suriname and French Guiana: Hahn *et al.* 2012; Northern Territory, Australia: Hahn *et al.* 2012; Jensen *et al.* 2013). However, other studies found little genetic differentiation between rookeries over larger areas. In the Indian Ocean, Shanker *et al*. (2004) detected no population subdivision along 2,000 km of east India coastline. In the east Pacific Ocean, rookeries in Costa Rica and Mexico, separated by more than 500 km, were not genetically distinct (Bowen *et al.* 1998; Hahn *et al.* 2012). Fine-scale population structure also was not found across 13 solitary and arribada nesting beaches along the Baja Peninsula and the main coast of Mexico (Rodríguez-Zárate *et al.* 2013). Genetic diversity was low among these nesting sites indicating a population collapse likely due to localized over exploitation (Rodríguez-Zárate *et al.* 2013). Lopez-Castro and Rocha-Olivares (2005) found genetic diversity in solitary nesting assemblages from the Baja California Peninsula to be significantly lower than arribada nesting populations along the east Pacific coast of Mexico and Costa Rica. They concluded that the genetic composition of the Baja population indicates reproductive isolation and genetic differentiation. They believed that the loss of genetic diversity and the differences in mating strategies distinguished the Baja population from the arribada beaches on the main continent, and recommended that the peninsular population be considered a distinct management unit (Lopez-Castro and Rocha-Olivares 2005). Wallace *et al.* (2010b) identified two distinct regional management units in the east Pacific (arribada and solitary nesters) based on a meta-analysis of genetic and other relevant data on olive ridley life history and biogeography. They believed the arribada and solitary nesting assemblages warranted separate management considerations, given there were differences in genetic diversity, trends, and abundance between the two types of nesting behaviors.

In the western Atlantic Ocean, Plot *et al.* (2012) found low genetic diversity in the French Guiana population. They felt the low diversity could be attributed to a recent (300,000 years ago) colonization of the western Atlantic by olive ridley turtles (Bowen *et al*. 1998), but was more likely indicative of a recent population collapse due to human over-exploitation (Plot *et al.* 2012).

A.2.3.1.5. Habitat Use or Ecosystem Conditions

Information is sparse about the condition of habitats and/or ecosystems and their impact on olive ridley populations. Olive ridleys occupy marine ecosystems that occur over vast areas and are considered nomadic in the eastern Pacific (Plotkin 2010). In this region, olive ridleys often associate with the highly productive area called the Costa Rica Dome located between 8 to 10°N and 88 to 90°W, which is characterized by a shallow (within 10 m of the surface) thermocline and areas of upwelled waters rich in prey items (Swimmer *et al.* 2009). Olive ridleys appear to forage throughout the eastern tropical Pacific Ocean, often in large groups, or flotillas, and are occasionally found associated with floating debris (Arenas and Hall 1992). Flotsam may provide the turtles with food, shelter, and/or orientation cues in the open ocean. Olive ridleys comprised the vast majority (75%) of sea turtle sightings associated with flotsam (Arenas and Hall 1992).

The El Niño Southern Oscillation, which is an irregular pattern of periodic variation between warm and cool sea surface temperatures, is probably the most significant ecosystem condition that may affect the survival status of olive ridleys in the eastern Pacific Ocean. The cool, nutrient rich and biologically productive waters characteristic of this region become warmer and less productive during an El Niño. This warming impacts lower trophic levels in the ocean (i.e., planktonic communities) and eventually, the upper trophic levels as well (i.e., nekton). Warming trends in the Pacific, caused by the frequent occurrence of El Niños since 1976, may be responsible for the decline in zooplankton in the California Current and the corresponding decline in higher trophic level vertebrates of this marine ecosystem (Hill 1995). The direct impact of El Niños on olive ridleys is unknown, but olive ridleys appear to change migration pathways in response to shifts in food availability during El Niños (Plotkin 2010**).** Because olive ridleys in the eastern Pacific are highly vagile, and seemingly adaptable to fluctuating environmental conditions, they possess the ability to shift from an unproductive habitat to one where the waters are biologically productive (Plotkin 1994, 2010).

Stable isotope analysis can complement satellite data of olive ridley movements and identify important foraging areas (reviewed by Jones and Seminoff 2013). Olive ridleys forage on a variety of marine organisms, including tunicates, gastropods, crustaceans, fishes, and algae (reviewed by Jones and Seminoff 2013). These prey show δ13C and δ15N isotopic values that reflect regional food webs, and olive ridleys retain these values in their soft tissue long after they depart the foraging area. In the eastern tropical Pacific, similar isotope signatures were found among and between juveniles and adults (Hess *et al.* 2008; Peavey *et al.* 2013) and sex class (Hess *et al.* 2008), indicating these groups fed on similar prey. Isotope values may also vary by region. Peavey *et al.* (2013) examined isotope signatures from olive ridleys captured in three regions: (1) Gulf of California (n = 29); (2) North Equatorial Current (n = 33); and (3) Eastern Pacific Warm Pool (n = 138). The Gulf of California isotope values were different from the other two regions, but no difference was found between the North Equatorial Current and the Eastern Pacific Warm Pool. They also found that δ13C and δ15N values decreased as distance from shore increased in the North Equatorial Current region, which can be explained by the fact that plants in pelagic areas have thicker cells and carbon uptake would take longer to assimilate in the turtle's tissue (Peavey *et al.* 2013). These studies help elucidate foraging strategies and habitat use.

A.2.3.1.6. Abundance and Population Trends

This section on the Endangered populations is organized by arribada and solitary nesting beaches. The available life history data, coupled with the genetic data, underscore the need to examine the status of solitary nesting populations independently from arribada nesting populations. As discussed earlier (see Section A.2.3.1.3. Demography), life history differences between solitary nesters and arribada nesters exist (Bernardo and Plotkin 2007; Kalb 1999; Plotkin and Bernardo 2003) that can impact population growth. Second, Lopez-Castro and Rocha-Olivares (2005) demonstrated genetic differences between the solitary and arribada nesting populations. Jensen *et al*. (2006) found a significant increase in multiple paternity (i.e., more than one male fertilizing eggs in a clutch) in nests from arribada beaches and attributed population size and the associated increase in male encounter rates as the major factor. These studies demonstrate that solitary and arribada mating systems are distinct from each other. Finally, many studies found lower nest success for arribada beaches compared to solitary beaches (e.g., Bézy *et al.* 2013), which may affect population growth.

Population abundance has been assessed and monitored, on the nesting beaches, using the standard survey method for sea turtles (Schroeder and Murphy 1999) where the number of female turtles observed nesting on the beach and/or their tracks left in the sand are counted during some pre-determined time interval and over a standard length of beach. Most olive ridley nesting beach surveys have taken place at arribada beaches where mass emergences in a spatially limited area present challenges to counting turtles directly or counting individual tracks left in the sand. Several methods have been used to estimate the number of turtles nesting during an arribada (Bézy and Valverde 2012; Cornelius and Robinson 1985; Gates *et al*. 1996; Márquez-M. and Van Dissel 1982; Valverde and Gates 1999). The olive ridley abundance estimates presented herein were derived from multiple methods at the different arribada beaches and in some cases the method used at a specific arribada beach has changed over the years (e.g, La Escobilla). This renders comparisons among arribada beaches problematic and discerning population trends over time complicated. A further complication is that many nesting population estimates from arribada beaches have been calculated as the sum total of all the turtles nesting during arribadas within a given nesting season. An individual olive ridley may nest on the same beach multiple times during a nesting season and thus the sum total of all the turtles or tracks counted during surveys is not directly equivalent to the number of turtles present in any given nesting population.

Arribada Beaches
Historically there were several large arribada nesting populations in Mexico (Table 1). These arribadas occurred at: Mismaloya, Tlacoyunque, Chacahua, La Escobilla, and Moro Ayuta. The current abundance of olive ridleys compared with former abundance at each of the large arribada beaches indicates the populations experienced steep declines due to over-exploitation (Abreu-Grobois and Plotkin 2008; Cliffton *et al*. 1982). The only exception is Ixtapilla, which was not discovered until 1994 (Abreu-Grobois and Plotkin 2008), and long-term nesting trends are unknown.

A recovery criterion is that females estimated to nest annually at source beaches must be stable or increasing over 10 years. Based on the current number of olive ridleys nesting in Mexico (Table 1), three populations appear to be stable (Mismaloya, Tlacoyunque, and Moro Ayuta), two increasing (Ixtapilla, La Escobilla) and one decreasing (Chacahua), but none of these populations have returned to their pre-1960s abundance (Abreu-Grobois and Plotkin 2008). Cliffton *et al*. (1982) derived a conservative estimate of 10 million adults prior to 1950. By 1969, after years of adult harvest, the estimate was just over one million. Olive ridley nesting at La Escobilla rebounded from approximately 50,000 nests in 1988 to over 700,000 nests in 1994 (Márquez-M. *et al*. 1996) and more than a million nests by 2000 (Márquez-M. *et al*. 2005). Abreu-Grobois and Plotkin (2008) estimated a mean annual estimate of 1,013,034 females nesting annually from 2001-2005 at La Escobilla. In 2009, over 350,000 nests were estimated during two arribadas, however data for the entire season are not available (Peralta and Peñaflores 2010 as cited in Ocana *et al.* 2012). The increases observed on the nesting beaches are supported by at-sea estimates of density and abundance, indicating a weighted average of the yearly abundance estimates of 1.39 million (Confidence Interval: 1.15 to 1.62 million) (Eguchi *et al*. 2007).

From the late 1990s through 2008, smaller females and fewer eggs per clutch were documented in Mismaloya, Mexico (Castellanos-Michel *et al.* 2008). New, smaller recruits to the nesting population may be a result of years of conservation efforts on the nesting beaches and indicate the population has stabilized (Castellanos-Michel *et al.* 2008).

Non-Arribada Beaches

In Mexico, olive ridleys nest more or less along the entire coastline, but the most concentrated area of nesting lies between the states of Sinaloa in the north to Chiapas in the south (R. Briseño, Banco de Información sobre Tortugas Marinas (BITMAR), personal communication, 2006; A. Abreu, Unidad Academica Mazatlan, personal communication, 2006). Elsewhere nesting is considered sporadic with the exception of Baja California Sur, where a small, solitary nesting population has been reported (Lopez-Castro and Rocha-Olivares 2005). Nest density varies along Mexico's coast: density is highest adjacent to arribada beaches and declines with increasing distance from arribada beaches (R. Briseño, BITMAR, personal communication, 2006; A. Abreu, Unidad Academica Mazatlan, personal communication, 2006). Nesting population trends for most beaches indicate they are stable or increasing. Stable beaches include: El Verde, Maruata-Colola, Puerto Arista, and Moro Ayuta. Increasing trends are reported for Platanitos and Cuyutlán (Abreu-Grobois and Plotkin 2008).

Table 1. Endangered populations of olive ridley arribada and solitary nesting beaches in Mexico, and estimates of annual abundance at each site and current trends ▲ = increasing; ▼ = decreasing; ▬ = stable; ? = unknown [*Note*: All sites are considered depleted from historical abundance except where noted].

Location	Years	Annual Number	Trend	References
ARRIBADA				
Mismaloya[1]	2001-2006	2,328 protected[2] nests	▬	Abreu-Grobois and Plotkin (2008)
Tlacoyunque[1]	1997	608 protected[2] nests	▬	Abreu-Grobois and Plotkin (2008)
Ixtapilla[3]	1999-2005	2,900-10,000 nests	▲	Abreu-Grobois and Plotkin (2008)
Chacahua[1]	2001-2005	2,042 nests	▼	Abreu-Grobois and Plotkin (2008)
La Escobilla	2001-2005	1,013,034 females	▲	Abreu-Grobois and Plotkin (2008)
Moro Ayuta[1]	2006	10,000 - 100,000 nests	▬	R. Briseño, BITMAR, and A. Abreu, Unidad Academica Mazatlan, pers. comms., 2006
SOLITARY				
El Verde	2000-2005	1,160 protected[2] nests	▬	Abreu-Grobois and Plotkin (2008)
Platanitos	2000-2005	1,301 nests	▲	Abreu-Grobois and Plotkin (2008)
Cuyutlán	1999-2003	1,257 nests	▲	Abreu-Grobois and Plotkin (2008)
Maruata-Colola	1999-2003	4,198 nests	▬	Abreu-Grobois and Plotkin (2008)
Puerto Arista	1999-2004	707 nests	▬	Abreu-Grobois and Plotkin (2008)
Moro Ayuta		No estimate available	▬	R. Briseño, BITMAR, and

Location	Years	Annual Number	Trend	References
				A. Abreu, Unidad Academica Mazatlan, pers. comms., 2006
Nuevo Vallarta	~2000-2010	~4,900 nests[4]	?	Maldonado-Gasca and Hart 2012
San Cristóbal	1995-2006	89 nests	?	Rodríguez *et al.* 2010
El Suspiro	1995-2006	220 nests	?	Rodríguez *et al.* 2010

[1] Large arribadas once occurred at these beaches but no longer do (Abreu-Grobois and Plotkin 2008; Cliffton *et al*. 1979; Hoekert *et al*. 1996).

[2] Protected nests are defined as those nests that would not be poached, predated, and otherwise lost (Abreu-Grobois and Plotkin 2008).

[3] Olive ridley nesting at this site was not recorded prior to 1994 (Abreu-Grobois and Plotkin 2008). It is unknown whether the population is depleted from historical abundance.

[4] Based on reported monitoring of 14 km of beach and nesting density of >350 nests/km/year (Maldonado-Gasca and Hart 2012).

As discussed earlier, at-sea abundance estimates appear to support an overall increase in the Endangered breeding colony populations on the Pacific coast of Mexico (Eguchi *et al*. 2007). At-sea estimates of density and abundance were determined from shipboard line-transect surveys conducted along the Mexico and Central American coasts during summer and autumn of 1992, 1998, 1999, 2000, 2003, and 2006 (Eguchi *et al*. 2007). A weighted average of the yearly estimates of olive ridley abundance was 1.39 million (Confidence Interval: 1.15 to 1.62 million), which is consistent with the increases seen on the eastern Pacific nesting beaches as a result of protection programs that began in the 1990s (Eguchi *et al*. 2007).

A.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 five listing factors (see below). In considering whether a species reclassification or delisting is warranted, we look at each factor singularly and in aggregate and whether these factors contribute to the extinction risk of the species. 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.

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

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). 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). These activities have increased in many parts of the olive ridley's range and pose threats to major nesting sites in Central America (Cornelius *et al*. 2007). However, data on specific impacts to habitat resulting from construction, beach armoring, etc., on the Endangered breeding colony populations in Mexico are lacking. In addition, coastal development is usually accompanied by artificial lighting. The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources, which may direct them away from the water (Witherington and Bjorndal 1991). Although empirical data on the impacts of destruction, modification, and curtailment of the olive ridley's habitat or range are lacking from many areas, habitat loss is likely given current human encroachment on coastal habitats (e.g., Honey and Krantz 2012). Coastal construction, pollution, and other human-related impacts to the olive ridley's habitat will likely increase as Mexico's population expands and tourism increases (e.g., Honey and Krantz 2012), which has the potential to negatively affect the availability of nesting habitat, as well as nesting success.

At sea, there are numerous potential threats including marine pollution, oil and gas exploration, lost and discarded fishing gear, changes in prey abundance and distribution due to commercial fishing, habitat alteration and destruction caused by fishing gear and practices, agricultural runoff, and sewage discharge (Frazier *et al*. 2007; Lutcavage *et al*. 1997). There are no empirical data to determine the impacts of these activities on olive ridley populations.

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 olive ridley 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 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 olive ridley prey distribution and abundance. However, olive ridleys in the east Pacific Ocean are highly vagile, and seemingly adaptable to fluctuating environmental conditions. They possess the ability to shift from an unproductive habitat to one where the waters are biologically productive, which may minimize the impacts of climate change (Plotkin 1994, 2010).

Sea-level rise from global warming is also a potential problem for areas with low-lying beaches where sand depth is a limiting factor. Soares *et al.* (2013) predict a 0.6 m sea level rise over the next 100 years for nesting beaches in Baja California Sur, Mexico, resulting in loss of habitat and inundation of olive ridley nests. 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. Dewald and Pike (2013) examined hurricane paths in the northwest Atlantic and northeast Pacific Oceans from 1970 through 2007 to quantify the frequency of impacts on sea turtle nesting sites. They found a slight, but significant increase in annual numbers of storms during the last 38 years (linear regression, $F_{1,36} = 3.98$, $R^2 = 0.10$, $P = 0.05$). Approximately 97% of nesting sites (all sea turtle species) in the eastern Pacific and western Atlantic Oceans were affected by hurricanes during this period. However, olive ridley nesting beaches were generally in areas exposed to only a few hurricanes over the last 38 years (Dewald and Pike 2013).

In summary, data are lacking on direct habitat loss of the olive ridley Endangered breeding colony populations on the Pacific coast of Mexico, but construction that adversely impacts coastal and estuarine habitat occurs and may increase in Mexico (e.g., Honey and Krantz 2012). The few studies on climate change impacts indicate the olive ridley may be more resilient on foraging grounds (Plotkin 2010) or not as exposed to severe weather on nesting beaches (Dewald and Pike 2013). However, other effects from climate change such as skewed sex ratios and high egg mortality likely will occur (see Section A.2.3.2.5). Presently most of the Endangered breeding colony populations appear to be stable or increasing since collapses due to over-exploitation, and at-sea estimates of density and abundance of the olive ridley indicate increasing numbers consistent with the increases seen on the eastern Pacific nesting beaches as a result of protection programs that began in the 1990s (see Section A.2.3.1.6 Abundance and Population Trends). As a result of this recent abundance data indicating increases or stabilization, the Services are cautiously optimistic that habitat destruction, modification, or curtailment of habitat or its range as a direct result of climate change no longer pose an immediate threat. However, the threats have not been eliminated and the Services remain concerned about the increased habitat loss due to human encroachment.

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

Olive ridleys and their eggs have been overutilized worldwide, including from the Endangered breeding colony populations on the Pacific coast of Mexico. The history of use and detailed accounts of this use are reviewed by Campbell (2007a), Cornelius *et al*. (2007), and Frazier *et al*. (2007).

The current impact of human use of olive ridley turtles and their eggs on populations is difficult to evaluate because there are many factors that contribute to a population's growth and decline (e.g., incidental take in commercial fisheries); however, Cornelius *et al*. (2007) identify several solitary nesting beaches and arribada beaches where current egg use is causing declines.

Large-scale egg use historically occurred at arribada beaches in Mexico, concurrent with the use of adult turtles at these beaches (Cliffton *et al*. 1982). The high level of adult mortality is believed to be the reason why rapid and large nesting population declines occurred in Mexico (Abreu-Grobois and Plotkin 2008; Cornelius *et al*. 2007).

The 1990 nationwide ban on harvest of nesting females and eggs has decreased the threat to the Endangered population in Mexico. The nesting population at La Escobilla, Oaxaca, Mexico, has increased from 50,000 nests in 1988 to more than a million nests in 2000 as a result of the harvest prohibitions and the closure of a nearshore turtle fishery (Cornelius *et al*. 2007). Over 1 million females are estimated to nest each year at La Escobilla (Abreu-Grobois and Plotkin 2008). However, illegal egg use is still widespread (Ocana 2010). Approximately 300,000-600,000 eggs were seized each year from 1995-1998 (Trinidad and Wilson 2000).

Olive ridleys were overutilized for commercial purposes in two legal turtle fisheries that operated in the eastern Pacific Ocean for over 20 years prior to a 1990 closure (Campbell 2007a; Cliffton *et al*. 1982; Green and Ortiz-Crespo 1982). The Mexican turtle fishery caused rapid, large declines at olive ridley arribada beaches in Mexico (Cliffton *et al*. 1982) that were so dramatic they have been widely referred to in the literature as population collapses, crashes, or extinctions. Genetic diversity is extremely low in these populations, indicating the populations collapsed (Rodríguez-Zárate *et al.* 2013). An estimated 75,000 turtles were taken each year for over two decades until 1990 when the fishery closed (Aridjis 1990). The fishery closure, along with the protection on the nesting beaches, is generally believed to have resulted in an increase in the population (Eguchi *et al.* 2007; Godfrey 1997; Márquez-M. *et al*. 1996; Pritchard 1997).

An Ecuadorian turtle fishery also existed during the 1970s and killed several hundred thousand olive ridleys during this time (Green and Ortiz-Crespo 1982). In 1978 alone, 80,535 to 89,483 olive ridleys were harvested (Green and Ortiz-Crespo 1982). This fishery is also believed to have contributed to the decline in the number of olive ridleys nesting on Mexican arribada beaches. A direct link between Mexico nesting beaches and Ecuadorian waters was established when olive ridleys, tagged while nesting in Mexico, were later captured in the Ecuadorian turtle fishery (Green and Ortiz-Crespo 1982).

The closure of the olive ridley turtle fishery in Mexico has decreased the threat to the population. However, illegal take of adult turtles still occurs in the region and the impact of this take is unknown. There is evidence that thousands of olive ridleys are still taken each year along the Pacific coast of Mexico (Frazier *et al*. 2007). The Mexican enforcement agency, Procuraduria Federal de Protección al Ambiente (PROFEPA), seized approximately 1.7 million turtle eggs, 1,900 units of turtle leather, and several hundred dead and live whole turtles from 1995-1998 in the State of Oaxaca (species not specified) (Trinidad and Wilson 2000).

Recreational, scientific, or educational overutilization has not been reported for olive ridleys.

In summary, despite ongoing illegal harvest, most of the Endangered breeding colony populations appear to be stable and some nesting beaches are increasing since collapses from over-exploitation. At-sea estimates of density and abundance of the olive ridley indicate increased numbers consistent with the increases seen on the eastern Pacific nesting beaches as a result of protection programs that began in the 1990s (see Section A.2.3.1.6 Abundance and Population Trends). The Services conclude that the threat from overutilization for commercial, recreational, scientific, or educational purposes has decreased substantially since 1990.

A.2.3.2.3 Disease or predation:

Little is known about disease in olive ridleys (George 1997). Nothing is known about the impact of disease on olive ridley abundance. The only disease identified in the literature thus far for olive ridleys is fibropapillomatosis, sometimes associated with a herpes-virus found in sea turtles nearly worldwide (Aguirre *et al.* 2000; Herbst 1994). The incidence of fibropapillomatosis is not believed to be high in olive ridleys. However, the disease has been observed in olive ridleys nesting in Mexico (Reséndez *et al.* 2010).

Predation on olive ridleys, their eggs, and offspring occurs on land and in the ocean throughout their range and the relative impacts of this mortality on nesting populations is unknown. Most studies were conducted on the Threatened population nesting sites (see Section B.2.3.2.3). Eggs and hatchlings fall prey to numerous mammalian, avian, reptilian, invertebrate, and fungal organisms. At La Escobilla, Mexico, beetles prey upon eggs and hatchlings (Harfush *et al.* 2008b). The effects of the beetle infestation at the population level is not fully understood and needs further evaluation (Abreu-Grobois and Plotkin 2008; Harfush *et al.* 2008b). In the ocean, sharks, billfish, whales, and birds may prey on adults (Frazier *et al*. 1994, 1995; Pitman and Dutton 2004) and hatchlings (Villasñor *et al.* 2010). The gut contents of a mahi mahi caught south of Mazatlan, Mexico, contained hatchlings in early digestive state indicating the hatchlings were likely eaten just off nesting beaches in the southern tip of Mazatlan (Villasñor *et al.* 2010).

In summary, disease is believed to be a relatively minor threat to the Endangered populations. The best available data suggest that current nest and hatchling predation at several nesting beaches and in water habitats is a potential threat but does not indicate severity or whether the population trend is affected by the predation rate.

A.2.3.2.4 Inadequacy of existing regulatory mechanisms:

The migratory nature of olive ridleys requires international collaboration to ensure their survival. For the purposes of the Endangered populations section, we consider instruments (e.g., regulations, treaties, conventions, agreements) that relate to the conservation and recovery of olive ridleys in the east Pacific Ocean, assuming the majority of efforts in the western Pacific Ocean and southeast Asia would affect turtles that originate from breeding colonies listed as Threatened (see Section B.2.3.2.4 for a global evaluation).

The conservation and protection of olive ridleys 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 of Mexico. In 1986, Mexico established 17 reserve areas to protect sea turtles. In 1990, Mexico banned the harvest and trade of sea turtles. Mexico requires the use of turtle excluder devices in their shrimp fishery to reduce sea turtle bycatch. Local community efforts are numerous. For example, the nongovernmental organization, Grupo Tortuguero, established 30 community sites for monitoring beaches and in-water surveys along the Baja Peninsula and Gulf of California (Esliman *et al.* 2012). In the state of Nayarit, Mexico, there are seven centers for Sea Turtle Protection and Conservation and two Sea Turtle Protection Camps covering nearly 80 km of nesting beaches (Maldonado-Gasca and Hart 2012).

The U.S. implemented several fisheries regulations that remain in effect to reduce sea turtle bycatch including olive ridleys. For example, all commercial fishermen in the U.S. who incidentally take a sea turtle during fishing operations must handle the animals with due care to prevent injury to live sea turtles, resuscitate, if necessary, and returned safely to the water. No sea turtles may be consumed, sold, landed, kept below deck, etc. The U.S. Hawaii-based longline fishery operating in the central Pacific also incidentally takes olive ridleys from the Endangered populations (NMFS 2008). Olive ridley interaction and mortality rates 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.

As a result of these international, national, and local efforts, many of the anthropogenic threats have been lessened. The ban on direct harvest resulted in stable or increasing nesting Endangered breeding colony populations on the Pacific coast of Mexico, although the Chacahua arribada beach continues to decline. Conservation measures to reduce incidental bycatch have benefited the Endangered populations; however, fisheries remain a concern. The lack of comprehensive and effective monitoring and bycatch reduction efforts in many fisheries operations still allows substantial direct and indirect mortality (see Section A.2.3.2.5).

Considering the worldwide distribution of olive ridleys, virtually every legal instrument that targets or impacts sea turtles is almost certain to cover olive ridleys. A summary of the main regulatory instruments from throughout the world that relate to the conservation and recovery of olive ridleys is provided below.

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 Pacific Ocean. 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.”

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 January 2014. While the Convention provides a framework within which broad conservation objectives may be pursued, it does not specifically address sea turtle conservation (Hykle 2002). Additional information is available at http://www.cbd.int/.

Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES)
Known as CITES, this Convention was designed to regulate international trade in a wide range of wild animals and plants. CITES was implemented in 1975 and had 180 Parties as of February 2014. The most recent Parties are Angola and Iraq who signed in 2013 and 2014, respectively. 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 olive ridley is listed on Appendix I of CITES as threatened with extinction and international trade is prohibited. Additional information is available at http://www.cites.org.

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 2007 sea turtle resolution (C-07-03) agreed to by the Inter-American Tropical Tuna Commission (IATTC) (http://www.iattc.org) encompasses most of the elements in the Western and Central Pacific Fisheries Commission, but only requires that parties to the agreement expeditiously undertake research to explore the use of circle hook/bait combinations to reduce sea turtle bycatch in longline fisheries. The IATTC has also developed a memorandum of understanding with the Inter-American Convention for the Protection and Conservation of Sea Turtles.

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.

In 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. 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 general concerns of effectiveness of domestic and intergovernmental authorities, the conservation and protection measures in this region appear to be effective as evidenced by stable and increasing olive ridley Endangered breeding colony populations in Mexico. At-sea estimates of density and abundance of the olive ridley indicate increased numbers consistent with the increases seen on the eastern Pacific nesting beaches as a result of protection programs that began in the 1990s (see Section A.2.3.1.6 Abundance and Population Trends). For these reasons, the Services believe that the inadequacy of existing regulatory mechanisms no longer pose an immediate threat to the olive ridley Endangered populations.

A.2.3.2.5 Other natural or manmade factors affecting its continued existence:

Several manmade factors affect olive ridleys in foraging areas and on nesting beaches. Two of these are truly global phenomena: climate change and fisheries bycatch. As stated earlier (Section A.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). Climate change will impact the ecosystems that sea turtles depend upon (e.g., Doney *et al.* 2012). 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 olive ridleys exhibit temperature-dependent sex determination (reviewed by Wibbels 2003, 2007), there may be a skewing of future cohorts toward a strong female bias since warmer temperatures produce more female embryos (Hawkes *et al*. 2009; Sifuentes-Romero *et al*. 2013). More importantly, elevated sand temperatures can result in almost zero hatch success (Sifuentes-Romero *et al*. 2013). The effects of global warming are difficult to predict, but changes in reproductive behavior (e.g., remigration intervals, timing and length of nesting season) may occur (reviewed by Hamann *et al.* 2013; Hawkes *et al.* 2009). 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).

Incidental capture in fisheries remains a serious threat in the eastern Pacific (Frazier *et al.* 2007) where olive ridleys aggregate in large numbers off nesting beaches (Kalb 1999; Kalb *et al*. 1995), but the information available is incomplete (Pritchard and Plotkin 1995, NMFS and FWS 1998). The incidental capture of olive ridleys in this region has been documented in shrimp trawl, longline, purse seine, and gillnet fisheries (Fahy 2011; Frazier *et al*. 2007). Incidental capture of sea turtles in shrimp trawls is a serious threat along the coast of Central America, with an estimated capture for all species of sea turtles exceeding 60,000, in one year (1993), most of which were olive ridleys (Arauz 1996). A study conducted off Costa Rica in two shrimp trawl fisheries estimated that over 15,000 sea turtles were taken annually off the Pacific coast of Costa Rica, with 90% of sea turtle species identified as olive ridleys with a mortality rate estimated at 38 percent (Arauz *et al.* 1998). Data from observers on U.S. and foreign large purse seine vessels have been gathered by the IATTC since the early 1990s, with sea turtle mortalities estimated between 17 and 172 per year, with olive ridleys comprising the majority, likely because they are proportionally more common than the other species (Fahy 2011). The numbers have dropped considerably since 2001, likely due to increased awarentess by fishermen and the passage of the IATTC resolution to mitigate sea turtle bycatch. Recent growth in the longline fisheries of this region is also a serious and growing threat to olive ridleys and has the potential to capture hundreds of thousands of ridleys annually (Frazier *et al*. 2007). Small scale fisheries operating in Peru using bottom set nets, driftnets, and longline fisheries were observed between 2000 and 2007. Almost 6,000 sea turtles were estimated to be captured annually, of which 240 were olive ridleys (Alfaro-Shigueto *et al.* 2011).

Other risk factors for sea turtles include interbreeding. Olive ridleys from the Endangered breeding colony populations on the Pacific coast of Mexico have been documented to interbreed with the green turtle (*Chelonia mydas*), and characteristics of neonates and embryos indicate hypridization (Hart *et al.* 2013); however, hybridization is not considered a threat to the continued existence of the olive ridley.

Increased exposure to heavy metals and other contaminants in the marine environment also affect olive ridleys (Bonzi *et al.* 2013). 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 hexachlorobenzenes and hexachlorohexanes had the lowest concentrations in samples taken from sea turtles (reviewed by Keller 2013).

In summary, the Endangered breeding colony populations appear to be stable and some nesting beaches are increasing since collapses due to over-exploitation. At-sea estimates of density and abundance of the olive ridley indicate increased numbers consistent with the increases seen on the eastern Pacific nesting beaches as a result of protection programs that began in the 1990s (see Section A.2.3.1.6 Abundance and Population Trends). Thus, threats from other natural and manmade factors have decreased as evidenced by stable and increasing olive ridley Endangered breeding colony populations in Mexico. However, incidental capture in fisheries and the impacts of climate change will pose a threat into the foreseeable future.

SUBSECTION B: THREATENED POPULATION

B.2.3.1 Biology and Habitat

B.2.3.1.1. Distribution

Globally, olive ridleys are found in coastal waters of over 80 countries (Abreu-Grobois and Plotkin 2008). The olive ridley has a circumtropical distribution, occurring in the Atlantic, Pacific, and Indian Oceans (Pritchard 1969). For distribution maps see State of the Worlds Sea Turtles OBIS-SEAMAP:http://seaturtlestatus.org. Olive ridleys nest in nearly 60 countries worldwide (Abreu-Grobois and Plotkin 2008). They do not nest in the United States. Key arribada beaches include La Flor in Nicaragua, Nancite and Ostinal in Costa Rica, La Marinera and Isla Cañas in Panama, Gahirmatha, Rushikulya, and Devi River in India, and Eilanti in Suriname. See Section B.2.3.1.6 Abundance and Population Trends (Table 2) for further details.

B.2.3.1.2. Migration

Regionally, olive ridleys may move between the oceanic zone and the neritic zone (Plotkin *et al*. 1995, Shanker *et al*. 2003a) or only occupy neritic waters (Pritchard 1976, Reichart 1993). Olive ridleys are not known to move between or among ocean basins. Thus, for the purposes of the Threatened population section, we examine movement within the Atlantic Ocean, Indian Ocean and western Pacific Ocean. See Section A.2.3.1.2 for discussion of migration for the eastern Pacific Ocean, which may include Threatened populations (e.g., Costa Rica, Panama).

Western Atlantic Ocean

In the western Atlantic, olive ridleys have been reported at sea as far north as the Grand Banks Region and as far south as Uruguay, encompassing a range between 43°N and 34°S (Foley *et al*. 2003; Fretey 1999; Stokes and Epperly 2006). However, they are most common in the waters of Guyana, Suriname, French Guiana, and Brazil and are not common elsewhere in the region. Female olive ridleys appear to remain in neritic waters during (Plot *et al*. 2012) and after breeding (Pritchard 1976; Reichart 1993). They forage on the continental plateau of Suriname and Guyana (Feuillet and de Thoisy 2007 as cited in de Boer 2013; Georges *et al*. 2008). There is little geographic overlap between the olive ridleys nesting in French Guiana/Suriname and those from Brazil (Godfrey and Chevalier 2004). Historic tag returns from females that nested in French Guiana/Suriname indicate that turtles migrate either south to foraging areas ranging from eastern Guyana to Amapa (Brazil), or north, to foraging areas ranging from the mouth of the Orinoco River to the islands of Trinidad and Tobago, and Margarita (Pritchard 1973; Schulz 1975). Tag returns from females that nested in Sergipe have been recovered in Sergipe or farther south in Brazil (Marcovaldi *et al*. 2000).

Eastern Atlantic Ocean
Information on olive ridleys in the eastern Atlantic is limited, but it is clear that olive ridleys are common throughout this region (Fretey *et al*. 2005). The species has been confirmed, or is thought to occur, along the coast between Mauritania and South Africa. The highest densities have been recorded in the Gulf of Guinea between the Ivory Coast and Gabon. Similar to the western Atlantic, there are few pelagic records of olive ridleys from the eastern Atlantic Ocean.

In the region, reproductively active males and females migrate toward the coast and aggregate at nearshore breeding grounds located near nesting beaches (Cornelius 1986; Hughes and Richard 1974; Kalb *et al*. 1995; Maxwell *et al.* 2011; Plotkin *et al*. 1991, 1996, 1997; Pritchard 1969). A significant proportion of the breeding also takes place far from shore (Pitman 1991, Kopitsky *et al*. 2000), and it is possible that some males and females may not migrate to nearshore breeding aggregations at all. Some males appear to remain in oceanic waters, are non-aggregated, and mate opportunistically as they intercept females *en route* to nearshore breeding grounds and nesting beaches (Plotkin 1994; Plotkin, *et al*. 1994, 1996; Kopitsky *et al*. 2000). During the internesting interval, females stayed in shallow waters (less than 50 m depth) within 30 km of the nesting beach in Gabon (Maxwell *et al.* 2011). Post-nesting females from Gabon and Angola travelled a minimum straight-line distance between 694 and 9,182 km within oceanic waters and largely in a southerly direction (Pikesley *et al.* 2013).

Indian Ocean
In the Indian Ocean, olive ridleys occur in the western ranges, but are seemingly uncommon. The species has been recorded in Oman, Mozambique, Tanzania, Kenya, Madagascar, and along the west coast of India (Table 2). Olive ridleys are most abundant in the northern Indian Ocean, particularly in the Bay of Bengal along the Indian coast.

As in the Atlantic Ocean, reproductively active males and females aggregate in large numbers nearshore during the breeding season. Tripathy (2013) found that males and females tended to aggregate within about a 60 km^2 area off Rushikulya, India. Mating pairs were sighted from 100 m to 8 km from shore, but the majority were sighted within 5 km of shore in front of the nesting beach. Kumar *et al.* (2010) observed similar large assemblages of breeding pairs in the 2006/2007 season off Rushikulya and described the assemblage as dynamic across the months and being spatially largest (25 km^2) during January and densest (68.1 turtles/ km^2 estimated at the surface) during March. Females nesting at Masirah Island, Oman, stayed within 59 km of the beach and shallow waters (< 40 m depth) during the interesting intervals (Rees *et al.* 2012).

Shanker *et al*. (2003a) tracked the migrations of a few post-nesting olive ridleys from India and found that the turtles moved almost randomly offshore in large circles before one turtle began a directed movement southwards. The seemingly meandering behavior may be due to the turtle's association with frontal regions between warm and cold core eddies in the Bay of Bengal (Shree Ram *et al.* 2009). Such behavior was similar to the non-directed movements of female olive ridleys in the east Pacific (Plotkin 2010; Plotkin *et al*. 1995).

Eight of nine post-nesting females from Masirah Island, Oman, travelled to four distinct neritic foraging grounds: (1) entrance to the Arabian Gulf; (2) coastal waters of western Pakistan; (3) Omani coastal waters north of Masirah; and (4) the Gulf of Masirah, which was also the internesting habitat for several of the females. One female, however, remained within Omani seas and moved randomly, spending a significant portion of time (39% of days with data) in oceanic waters deeper than 200 m (Rees *et al.* 2012).

Western Pacific Ocean

In the western Pacific, olive ridleys typically occur in tropical and warm temperate waters from Australia through southeast Asia. Although uncommon, olive ridleys are found in waters off mainland China and associated islands (Chan *et al.* 2007). Females nesting at Turtle Melville Island, Australia stayed within 17 to 37 km offshore during the nesting season, but travelled between 180-1,050 km to foraging areas after the nesting season ended (Whiting *et al.* 2007a). Their migration routes varied, but all stayed within 50-240 km of the coast.

Eastern Pacific Ocean

See Section A.2.3.1.2. Migration

B.2.3.1.3. Demography

See Section A.2.3.1.3. Demography

B.2.3.1.4. Taxonomy, Phylogeny, and Genetics

See Section A.2.3.1.4. Taxonomy, Phylogeny, and Genetics

B.2.3.1.5. Habitat Use or Ecosystem Conditions.

Western Atlantic Ocean

In the western Atlantic Ocean, little information exists on olive ridley habitat use. Females appear to remain in neritic waters during (Plot *et al*. 2012) and after breeding (Pritchard 1976; Reichart 1993). Females nesting in French Guiana and Suriname foraged over the continental shelf (Plot *et al.* 2012). In 2012, an olive ridley of unknown age class was observed swimming amongst floating *Sargassum* mats in offshore waters (depth 1,322 meters) of Suriname (de Boer 2013).

Eastern Atlantic Ocean

In the eastern Atlantic Ocean, post-nesting females from Gabon and Angola foraged within oceanic waters within approximately 200 km of the coast, off the continental shelf, where water depths were < 2000 m, with highest densities of olive ridley occurring in association with persistent fronts within the Angolan Exclusive Economic Zone (Pikesley *et al.* 2013). The warm Angolan current from the north converges with the cool Benguela current from the south in this area creating highly productive waters (Pikesley *et al.* 2013).

Indian Ocean

In the Indian Ocean, olive ridleys are most abundant in the northern region, particularly in the Bay of Bengal along the Indian coast. Little is known about the habitats that olive ridleys occupy in this part of their range. Large numbers aggregate nearshore during the breeding season, but their habitat use beyond the reproductive area is not well documented.

Western Pacific Ocean
In the western Pacific Ocean, post-nesting females in Australia exhibit plasticity by foraging in both coastal and oceanic habitat. Whiting *et al*. (2007a) reported females to forage in areas on or near the shallow coastal, continental shelf and slope in the Gulf of Carpentaria, Joseph Bonaparte Gulf, Cobourg Peninsula and the shelf-edge in the Northern Territory. Females stayed within foraging areas—one turtle spent 40 weeks in an area less than 150 km^2, and multiple turtles overlapped in the areas they foraged. McMahon *et al.* (2007) reported three of four females foraged in relatively deep water (> 100 m) with the maximum dive lasting 3.33 + 0.33 h, indicating long dives towards the seabed. Benthic foraging far from shore may be a unique foraging strategy among sea turtles (McMahon *et al.* 2007). However, foraging and diving behavior is not as well studied in olive ridleys compared to other species (Hochscheid 2014).

Eastern Pacific Ocean
See Section A.2.3.1.5. Habitat Use or Ecosystem Conditions

B.2.3.1.6. Abundance and Population Trends

This section on the Threatened populations is organized by arribada and solitary nesting beaches. As discussed earlier (see Section A.2.3.1.6.), the available life history data, coupled with the genetic data, underscore the need to examine the status of solitary nesting populations independently from arribada nesting populations. Table 2 provides information on nesting beach locations and, where known, estimates of abundance and trends.

As discussed earlier (see Section A.2.3.1.6), several methods have been used to estimate the number of turtles nesting during an arribada (Bézy and Valverde 2012; Cornelius and Robinson 1985; Gates *et al*. 1996; Márquez-M. and Van Dissel 1982; Valverde and Gates 1999). The olive ridley abundance estimates presented herein were derived from multiple methods at the different arribada beaches and in some cases the method used at a specific arribada beach has changed over the years (i.e., Ostional, Costa Rica). This renders comparisons among arribada beaches problematic and discerning population trends over time complicated.

Western Atlantic Ocean Arribada Beaches
In the western Atlantic Ocean, Suriname/French Guiana is considered an arribada nesting population. However, solitary nesting occurs in the region, and a sizeable nesting population occurs in Brazil (Table 2). Survey effort has fluctuated over the years and it is difficult to estimate recent abundance because of incomplete surveys during many years. Moreover, because the coastline of Suriname and French Guiana is dynamic, long-term surveys are difficult because the turtles change nesting locations frequently. We do know however that the Suriname olive ridley population is currently small and has declined by more than 90% (Abreu-Grobois and Plotkin 2008; Hoekert *et al*. 1996, Marcovaldi 2001). Schulz (1975) reported 3,290 olive ridley nests in 1968. By 1980, there were 1,080 olive ridley nests recorded in Suriname (Reichart and Fretey 1993). In 2005, only 138 nests were estimated to occur at Eilanti beach in the Galibi Nature Reserve, Suriname (Abreu-Grobois and Plotkin 2008).

In French Guiana, olive ridleys are known to nest on the western beaches and, in the last decade, were discovered nesting on eastern beaches (Kelle *et al*. 2004). The mean annual number of olive ridley nests was estimated to be 1,716 and 3,257 between 2002 and 2007 (Kelle *et al.* 2009). The mean annual number of nests on Cayenne Peninsula beaches was 2,105 (± 284) between 2002 and 2010 and significantly increased during that period (Plot *et al.* 2012). Between 2002 and 2008, arribada size at Cayenne averaged 104 ± 11 nests and the largest arribadas occurred in 2008 with 302 and 319 females emerging over two nights (Plot *et al.* 2012). It is unknown whether the apparent increase in nests represents turtles that relocated from Suriname, or inconsistent monitoring especially in the eastern region (Marcovaldi 2001), or a true population increase (Kelle *et al.* 2009). Based on genetic analysis, Plot *et al.* (2012) estimate the ancestral population to be 130 times larger than the current population. They estimated the ancestral population was about 20,000 breeding turtles, while the effective current population is between 100 and 150 breeding animals.

Eastern Atlantic Ocean Arribada Beaches
Arribada nesting has not been reported for this region (Table 2).

Indian Ocean Arribada Beaches
In the Indian Ocean/Bay of Bengal, three arribada beaches have been reported in the Indian State of Odisha (formerly known as Orissa) (Pandav *et al*. 1998): Gahirmatha, Devi River mouth, and Rushikulya (Table 2). Nesting beach surveys at Gahirmatha have been conducted since the mid-1970s. Long-term data for the two other arribada beaches are unavailable. Survey effort on India beaches has fluctuated over the years and the methods used to census the nesting populations have also changed. As a result, the accuracy of estimates of population size are unclear, with estimates exceeding 700,000 turtles nesting in one arribada.

Evidence suggests that olive ridleys in this region have changed their nesting behavior. Since recordkeeping began at Gahirmatha in the 1970s until the mid-1990s, there have been two arribadas recorded there during each nesting season. On rare occasions, there was only one arribada or no arribada recorded during a nesting season. From the mid-1990s through the early 2000s, only one arribada at Gahirmatha was recorded annually (Shanker *et al*. 2003b). However in 2010, two arribadas occurred (Behera *et al.* 2012). Shanker *et al*. (2003b) compiled all of the available census data from the arribada beaches in India, derived a consensus estimate for each arribada, and then determined nesting population trends at Gahirmatha. From 1974 to 2001, at least one arribada in excess of 100,000 turtles occurred in most years at Gahirmatha, as well as smaller arribadas less than 1,000. In their revised estimates, Shanker *et al*. (2003b) took into account the fact that the same turtles nest in successive arribadas and that the same turtles nest at different arribada beaches, an important fact that had been overlooked in previous estimates of nesting population size. The most recent reliable abundance estimate for Gahirmatha during the 1999 arribada is approximately 180,000 nesting females. Long-term data for Gahirmatha indicate that the olive ridley nesting population increased during the 1980s, followed by a decrease during the 1990s (Shanker *et al*. 2003b). However, the decline was not significant, but Shanker *et al*. (2003b) concluded that the olive ridley nesting population may be declining or on the verge of decline.

Estimates of arribada size at Devi River mouth and Rushikulya are quite large and considered unreliable (Kumar *et al.* 2013; Shanker *et al*. 2003b). For example in 2004, only 23,561 (±2,326) olive ridleys were estimated to nest over four nights during an arribada at Rushikulya

(Tripathy 2008). In 2009 and 2010, an estimated 172,407 (±7,509) and 134,478 (±6,204) nested, respectively, at Rushikulya (Kumar *et al.* (2013). Considerable fluctuation in the estimates of the number of nesting females was also recorded in the 1990s, where estimates ranged from 8,000 to 200,000 (Pandav *et al.* 1994, 1998). The combined arribada beaches (Gahirmatha, Rushikulya, and Devi River) are considered to be stable over three generations (defined as 20 years per generation; Abreu-Grobois and Plotkin 2008).

Western Pacific Ocean Arribada Beaches
Arribada nesting is not reported for this region (Table 2).

Eastern Pacific Ocean Arribada Beaches
In the eastern Pacific Ocean, Threatened populations of the olive ridley nest south of Mexico to Colombia. Within this range lie several beaches where arribadas reportedly occurred in the past but no longer do, as well as beaches where they still occur: five in Nicaragua, two in Costa Rica, and one in Panama (Table 2). Current estimates for some of the beaches are either unavailable or are based on sporadic nesting beach surveys.

In Nicaragua, two arribada beaches still exist (Ruiz 1994): Playa Chacocente (located in the Chacocente Wildlife Refuge) and Playa La Flor (located in a private wildlife refuge). Hope (2002) combined data from Playa Chacocente and Playa La Flor for a mean arribada size of 66,885 and a frequency of five to seven arribadas per year from 1993 through 1999. In 1993, an estimated 27,427 olive ridley nests were laid during six arribadas at Playa La Flor (Ruiz 1994). In 2003 and 2004, 69,765 and 68,753 nests, respectively, were estimated at Playa La Flor (Honarvar and van den Berghe 2008). For the 2008 and 2009 nesting seasons, 27,947 females were estimated to nests in nine arribadas at Playa Chacocente and 521,440 females in eight arribadas at Playa La Flor (Gago *et al.* 2012). During the 2009 and 2011 nesting seasons at Playa Chacocente and Playa La Flor, a total of 4,146,986 hatchlings were estimated to be produced (Salazar *et al.* 2013). Population trends for Playa Chacocente are unknown. The nesting population at Playa La Flor is thought to be depleted but stable (Abreu-Grobois and Plotkin 2008).

In Costa Rica, there are two arribada beaches: Nancite Beach (located in the Santa Rosa National Park, Guanacaste Conservation Area) and 90 km to the south, Ostional Beach/Wildlife Refuge (located within the Tempisque Conservation Area on the Nicoya Peninsula).

A small and declining nesting population exists at Nancite Beach. In the early 1980s, large arribadas occurred at Nancite nearly monthly (Cornelius and Robinson 1982). In 1981, Cornelius and Robinson (1982) estimated that over 400,000 olive ridleys nested at Nancite during 11 arribadas that took place between April and November. A significant decline in the population size occurred during the 1970s, 1980s, and 1990s and the frequency of arribadas also decreased (Fonseca *et al.* 2009, 2010; Valverde *et al*. 1998). The number of females nesting decreased 42% between 1971 and 1984, 84% between 1971 and 1992, and 90% between 1971 and 2007 (Fonseca *et al.* 2009). However, more hatchlings were produced in the 2007 arribadas because density-dependent factors that lower hatchling production (e.g., nests destruction, nutrient loads) were less a factor. For 2007, an estimated 63-97% of nests produced a total of

300,124 hatchlings compared to 27% of nests with a total of 134,955 hatchlings in 1984 (Fonseca *et al.* 2009).

Nesting at Ostional, Costa Rica, has been reported to be increasing (Abreu-Grobois and Plotkin 2008), but also declining in more recent years (Valverde *et al.* 2012). Within the Ostional Wildlife Refuge, olive ridleys gather *en masse* on Ostional Beach and to the immediate south onto Nosara Beach. Arribadas occur there throughout the year, with the largest number of olive ridleys nesting between July and January. Monitoring in the Refuge began in the 1970s. At least four different census methods have been used since then to estimate the number of turtles nesting during an arribada: (1) visual counts of all turtles, (2) Cornelius and Robinson method (quadrat method), (3) Valverde and Gates method (strip transect in time method that counts only nesting turtles), and, more recently, (4) Chaves and Morera method (a modified strip transect in time method that counts all turtles on the beach, not just nesting turtles). Since 1980, the frequency of arribadas has increased, the area of the beach used during arribadas has increased, and the number of turtles nesting per arribada has increased (Chaves *et al*. 2005). The average arribada size in the main nesting beach increased from 75,000 turtles in 1980 to 125,000 turtles in 2003 (Chaves *et al*. 2005). The number of arribadas per year ranged from 7 to 16 and averaged 11.17 ± 2.29 (Chaves *et al*. 2005). From 2006-2010, arribadas ranged between 3,564 to 476,550 nesting females using the Valverde and Gates method (Valverde *et al.* 2012).

In Panama, olive ridley arribadas occur at Isla Cañas, part of the Panama National Wildlife Refuge system and Las Marinera. Historical data reported to the Inter-American Sea Turtle Convention for 1997-2000 is 60,000 nesting females a year at Isla Canas and 225 nests at La Marinera in 2013. These data are estimates and not based on standardized surveys so trends are unknown. Previous reports by Abreu and Plotkin (2008) indicated that 8,768 females nest each year and the population was declining.

Western Atlantic Ocean Non-Arribada Beaches
In the Atlantic Ocean, low-density nesting occurs in Guyana, Suriname, and French Guiana (Godfrey and Chevalier 2004, Kelle *et al.* 2004; Reichart 1993; see data discussed above in the arribada section). Whether these turtles are true arribada nesters (i.e., emerge synchronously) or solitary nesters is undocumented and thus it is difficult to differentiate between them. Numbers presented in the above West Atlantic Ocean section for arribada beaches therefore reflect the combined numbers of olive ridleys nesting on arribada beaches and non-arribada beaches along this coastline with the exception of Guyana where fewer than five nests were recorded annually from 2002 to 2006 in Guyana (L. Kelle, WWF, personal communication from Guyana Marine Turtle Conservation Society) and may now be extirpated (de Thoisy *et al.* 2010). In Brazil, olive ridleys nest in the states of Sergipe and Bahia. The nesting populations are small (Marcovaldi 2001), but increasing (da Silva *et al.* 2007; Godfrey and Chevalier 2004). The number of nests has increased from 100 nests in 1989/1990 (Godfrey and Chevalier 2004) to 252 nests in 1991/1992 and to 2,606 in 2002/2003 (da Silva *et al.* 2007).

Eastern Atlantic Ocean Non-Arribada Beaches
Widespread, low density olive ridley nesting occurs along many West African beaches generally from Gambia south to Angola (Barnett *et al*. 2004, Barbosa *et al*. 1998, Beyer 2002, Doussou Bodjrenou *et al*. 2005, Fretey *et al*. 2005, Hoinsoude *et al*. 2003, Gomez *et al*. 2003). In 2011,

an olive ridley nest was confirmed in the Langue de Barbarie National Park, Senegal, at approximately 15.99° N. latitude, the northernmost record for the olive ridley in the east Atlantic Ocean (Fretey *et al.* 2012). In Bioko, Equatorial Guinea, surveys over two nesting seasons (1996/1997 and 1997/1998) found 57 to 84 nests (Tomás *et al.* 2010). In Orango National Park, Guinea-Bissau, during 2 nesting seasons from 1992 through 1994, annual nest estimates ranged from 170 to 620 (Catry *et al.* 2009). At Palmeirinhas beach, Angola, an average 123 nests were counted each nesting season from 2003-2006 (Weir *et al.* 2007). In the Republic of Congo, annual nest numbers have dropped by about 50% from approximately 600 nests in 2003/2004 to less than 300 nests in 2009/2010 (Girard and Breheret 2013). In the Democratic Republic of Congo, 102 nests were recorded during the 2012-13 nesting season (Mbungu Ndamba 2013).

Indian Ocean Non-Arribada Beaches

In the Indian Ocean, widespread, low-density olive ridley nesting occurs in the western and northern region. The species has been recorded nesting in low numbers in Oman (Ross and Barwani 1995), Mozambique (Pritchard 1979), Tanzania (Frazier 1976), Kenya (Church 2005; low but increasing numbers (Oman 2013)), Madagascar (Pritchard 1979), and along the southwest coast of India (Krishna 2005). Olive ridley nesting is most concentrated in the northern Indian Ocean, particularly along the shores of the Bay of Bengal on the East Indian coast and Sri Lanka (Amarasooriya and Jayathilaka 2002; Tripathy *et al*. 2003). Raja Sekhar (2013) found nest densities as high as 34 nests/km on beaches adjacent to the Godvari River mouth in Andhra Pradesh, India. Abundance estimates and population trends are generally unavailable for most of this region. Declines of olive ridleys have been recorded in Bangladesh (Islam 2002; Sarker 2005), Myanmar (Lwin 2009; Thorbjarnarson *et al*. 2000), Malaysia (Limpus 1995), Pakistan (Asrar 1999), and southwest India (Krishna 2005). In Eritrea, more than 120 islands and coastal sites were surveyed between 2000 to 2008, and the first record of an olive ridley nest was reported for the Red Sea (Mebrahtu 2013).

Western Pacific Non-Arribada Beaches

In Indonesia, olive ridleys nest on beaches in the West Papua Province (known as the Manokwari region), and number of nests recorded from 2008 through 2011 ranged from 53 to 236, however survey effort was limited and likely not consistent across years (Suganuma *et al.* 2012). On Jamursba-Medi beach, on the northern coast of West Papua, 77 olive ridley nests were documented from May to October 1999 (Teguh 2000). Extensive hunting and egg collection, in addition to rapid rural and urban development, have reduced nesting activities in this area. On Hamadi beach, Jayapura Bay, in June 1999, an estimated several hundred ridleys were observed nesting. At Alas Purwo National Park, located at the eastern-most tip of East Java, olive ridley nesting was documented from 1992-1996. Recorded nests were as follows: from August to September 1993, 101 nests; between March and October 1995, 162 nests; and between April and June 1996, 169 nests. From these limited data, no conclusions could be reached regarding population trends (Suwelo 1999); however, Dermawan (2002) reports that there were up to 250 females nesting at this site in 1996, with an increasing trend.

In Malaysia, olive ridleys nest on the eastern and western coasts; however, nesting has declined rapidly in the past decade. The highest density of nesting was reported to be in Terrenganu, Malaysia, and at one time yielded 240,000 eggs (2,400 nests, with approximately 100 eggs per nest) (Siow and Moll 1982 as cited in Eckert 1993), while only 187 nests were reported from the area in 1990 (Eckert 1993), and were virtually extirpated by 1999 (Chan 2006). In eastern

Malaysia, olive ridleys nest very rarely in Sabah and in low numbers (Basintal 2002), and only a few records are available from Sarak (Eckert 1993).

In Australia, olive ridley nesting is scattered throughout northern Australia, with an estimated few thousand females nesting annually (Limpus 2008; Whiting *et al.* 2007b). The breeding population in northern Australia may be the largest population remaining in the southeast Asia and western Pacific region, although a full evaluation of their distribution and abundance is needed (Limpus 2008). Nesting also occurs in very low numbers on the west coast of Australia (Prince *et al.* 2010).

Eastern Pacific Non-Arribada Beaches

In Guatemala, there is widespread, low-density olive ridley nesting. The most current estimate available indicates there were over 2 million olive ridley eggs laid on the coast of Guatemala in the late 1990s (Muccio 2000). If we assume that the average clutch size is 100 eggs, then this represents approximately 20,000 nests. Higginson (1989) provided estimates from data collected by Ramboux (1982) and Rosales Loessener (1987) and stated that 21,067 olive ridleys nested during 1981-1982. It is unknown if this estimate refers to the number of nests laid or if it refers to nesting females. Empirical population trend data are unavailable for Guatemala, but olive ridleys are reported to be declining (Juarez and Muccio 1997). Muccio (1999) reported that solitary nesting ridleys are estimated to have declined 34% between 1981 and 1997.

In El Salvador, there is low-density olive ridley nesting. There is no current estimate available of the number of olive ridleys nesting along the coast of El Salvador. Population trend data are unavailable; however, the olive ridley nesting population was considered to be declining in 1989 (Formia *et al*. 2000). In addition, coastal residents in El Salvador are convinced that sea turtle populations are steadily declining (Arauz 2000).

In Honduras, there is widespread, low-density olive ridley nesting on the shores of the Gulf of Fonseca. Lagueux (1989) reported nesting occurs on 46 different Honduran beaches. In Punta Raton, Lagueux (1989) reported 742 nests from July through December 1987. There is no current estimate of the number of olive ridleys nesting along the coast of Honduras, and population trend data are unavailable.

In Nicaragua, there is widespread, low-density olive ridley nesting. There is no current estimate of the number of olive ridleys nesting on non-arribada beaches along the coast of Nicaragua, and population trend data are unavailable.

In Costa Rica, there is widespread, low-density olive ridley nesting. There is no current estimate of the number of olive ridleys nesting on non-arribada beaches along the coast of Costa Rica, and population trend data are unavailable. However, there are a few non-arribada beaches where data have been collected. These beaches include: San Miguel, Playa Caletas, Punta Banco, and Osa Peninsula. From 1998 through 2004, on average, 180 nests were documented in San Miguel. For Playa Caletas, 71 olive ridley nests were documented during the 2002–2003 nesting season; however, there were 226 unconfirmed events, most of which were believed to be olive ridleys. From 1996–2005, over 1,000 olive ridley nests were located to hatcheries and protected from predation and poaching. Punta Banco has been monitored since 1996 (Gaos *et al*. 2006). A declining trend in the number of nests (*note:* the trend includes hawksbills (*Eretmochelys imbricate*) and green turtles (*Chelonia mydas*), but these species only laid a few nests each year) laid there has been reported (Gaos *et al*. 2006). During the 1993-1994 nesting season on the Osa Peninsula, 3,155 olive ridley nests were recorded (Drake 1996).

In Panama, Cornelius (1982) reported that sea turtle nesting was widespread and that large nesting aggregations once occurred on at least 30 beaches. By the early 1980s, turtles had declined and were nesting in smaller aggregations on only 12 beaches (Cornelius 1982). The sea turtle species nesting in these aggregations were not reported and may represent other species as well as olive ridleys. Widespread, low-density olive ridley nesting still occurs in Panama. There is no current reliable estimate of the number of olive ridleys nesting on non-arribada beaches along the coast of Panama, and population trend data are unavailable. Cornelius (1982) reported that by the late 1970s and early 1980s, olive ridley abundance in Panama was lower compared to former abundance levels. R. Chang (cited personal communication in NMFS and FWS 1998) estimated 10,000 solitary ridleys nested annually throughout Panama (exclusive of Isla Cañas).

In Colombia, low-density nesting occurs, principally in the Playon de El Valle (Choco Region) and Parque Snaguianga in the south (Narino Department) (Amorocho *et al*. 1992; D. Amorocho, MONASH University personal communication 2007). During 2003-2007, 25 olive ridleys nests were documented on Parque Gorona, a small 1.2 km island in the south (D. Amorocho, MONASH University, personal communication 2007). Amorocho (1994) reported olive ridley nesting on Playa Larga but did not provide the numbers of turtles or nests. On another beach, La Cuevita, Martinez and Paez (2000) reported 112 olive ridley nests in 1998.

In Ecuador, although common in nearshore waters, olive ridleys had not been recorded to nest on Ecuadorian beaches. In 2004, a single nest was identified as an olive ridley nest based on an examination of a late-stage embryo (Alava *et al.* 2007).

In Peru, nesting is rare and only one or two nests have been recorded (Hays-Brown and Brown 1982; Kelez *et al.* 2009).

Table 2. Threatened olive ridley arribada and solitary nesting beaches and estimates of abundance expressed as arribada size, nests, or females at each site and trends ▲ = increasing; ▼ = decreasing; ▬ = stable; ? = unknown. See text in 'Abundance and Population Trends' for greater detail.

Country	**Beach**	**Years**	**Annual Numbers**	**Trend**	**References**
ARRIBADA					
Western Atlantic Ocean					
Suriname	Galibi Nature Reserve[1]	1995	335 nests	▼	Hoekert *et al*. 1996
French Guiana	Cayenne Peninsula	2002-2010	2,015 ± 284 nests	▲	Plot *et al.* 2012
Indian Ocean					
India	Gahirmatha, Devi River, Rushikulya	1990-2008	150-200,000 females	▬	Abreu-Grobois and Plotkin 2008

Country	Beach	Years	Annual Numbers	Trend	References
East Pacific Ocean					
Nicaragua	Chacocente	2008-2009	27,947 females	?	Gago *et al.* 2012
Nicaragua	La Flor	2008-2009	521,440	▬	Gago *et al.* 2012
Nicaragua	Masachapa		No estimate available	?	Cornelius 1982; Margaritoulis and Demetropoulous 2003
Nicaragua	Pochomil		No estimate available	?	Cornelius 1982; Margaritoulis and Demetropoulous 2003
Nicaragua	Boquita		No estimate available	?	Cornelius 1982[2]
Costa Rica	Nancite	1999-2007	256-41,149 turtles per arribada	▼	Fonseca *et al.* 2009
Costa Rica	Ostional	2006-2010	3,564-476,550 turtles per arribada	▲[3]	Valverde *et al.* 2012
Panama	Isla Cañas	2006	8,768 turtles per year	▼	Abreu-Grobois and Plotkin 2008
SOLITARY					
Western Atlantic Ocean					
Suriname			No estimate available	?	Godfrey and Chevalier 2004; Kelle *et al.* 2004
Guyana			No estimate available	?	Godfrey and Chevalier 2004; Kelle *et al.* 2004
French Guiana			No estimate available	?	Godfrey and Chevalier 2004; Kelle *et al.* 2004
Brazil	Sergipe	2002-2003	2,606 nests	▲	da Silva *et al*. 2007
Eastern Atlantic Ocean					
Gambia			No estimate available	?	Barnett *et al*. 2004
Guinea Bissau	Orango National Park	1992-1994	170-620 nests	?	Catry *et al.* 2009
Sierra Leone			No estimate available	?	Siaffa *et al*. 2003
Ivory Coast			No estimate available	?	Gomez *et al*. 2003
Ghana			No estimate available	?	Beber 2002, 2008
Togo			No estimate available	?	Hoinsoude *et al*. 2003
Benin			No estimate available	?	Doussou Bodjrenou *et al*. 2005
Boiko, Säo Tome, Corisco, Mbanye, Hoco Islands			No estimate available	?	Fretey *et al*. 2005
Cameroon			No estimate available	?	Fretey *et al*. 2005
Equatorial Guinea		1996-1998	57-84 nests		Tomás *et al.* 2010

Country	Beach	Years	Annual Numbers	Trend	References
Gabon			No estimate available	?	Fretey *et al*. 2005
Republic of Congo		2003-2010	300-600 nests	▼	Girard and Breheret 2013
Democratic Republic of Congo		2012 - 2013	102 nests	?	Mbungu Ndamba 2013
Angola	Palmeirinhas	2003-2006	123 nests	?	Weir *et al.* 2007
Liberia			No estimate available	?	E. Possardt, FWS, pers. comm., 2007
Indian Ocean					
Mozambique			No estimate available	?	Pritchard 1979
Madagascar			No estimate available	?	Pritchard 1979
Kenya	Watamu			▲[4]	Oman 2013
Tanzania			No estimate available	?	Frazier 1976
Oman	Masirah Island	1977	150 females	?	Ross and Barwani 1995
India	Entire east & west coasts		No estimate available	?	Behera and Kar 2013; Krishna 2005;Tripathy *et al*. 2003
Union Territory of India	Andaman & Nicobar Islands	2001	185 nests	▼	Abreu-Grobois and Plotkin 2008
Pakistan	Hawkes Bay	1996-1997	2 nests	▼	Abreu-Grobois and Plotkin 2008
Sri Lanka	Northwest, west & southern coasts		No estimate available	?	Amarasooriya and Jayathilaka 2002
Bangladesh	St. Martin's	2001	7 females	▼	Abreu-Grobois and Plotkin 2008
Myanmar		1999	700 nests	▼	Abreu-Grobois and Plotkin 2008
Thailand	Thaimaung, Pharathong Island, Maikaw Beach	1996-2000	30 nests	▼	Abreu-Grobois and Plotkin 2008
Western Pacific Ocean					
Australia	Northern, northeast, & western beaches		No estimate available	?	Limpus 1975, 2008; Prince *et al.* 2010; Whiting 1997b
Brunei			No estimate available	?	Shanker and Pilcher 2003
Malaysia	Terengganu	1998-1999	10 nests	▼	Abreu-Grobois and Plotkin 2008
Indonesia	Alas Purwo	1993-1998	230 nests	▲	Abreu-Grobois and Plotkin 2008
Indonesia	Jamursba-Medi		No estimate available	?	P. Dutton and M. Tiwari, NMFS, pers. comms. 2007; Teguh

Country	Beach	Years	Annual Numbers	Trend	References
					2000
Vietnam			No estimate available	?	Shanker and Pilcher 2003
East Pacific Ocean					
Guatemala	Hawaii Beach & others	2005	1,004 females[5]	▼	Abreu-Grobois and Plotking 2008
El Salvador	Toluca, San Diego & others		No estimate available	?	Hasbún and Vasqúez 1999
Honduras	Punta Raton and others		No estimate available	?	Lagueux 1991
Nicaragua	Entire Pacific coast		No estimate available	?	Pritchard 1979
Costa Rica	Entire Pacific coast		No estimate available	?	Pritchard 1979
Panama			No estimate available	?	Pritchard 1979
Colombia	La Cuevita		No estimate available	?	Martinez and Paez 2000; Ramírez-Gallego and Barrientos-Muñoz 2012
Ecuador	Manta		No estimate available	?	Alava *et al.* 2007

[1] Large arribadas once occurred at these beaches but no longer do (Cliffton *et al*. 1979, Hoekert *et al*. 1996).
[2] Masachapa, Pochomil, and Boquita were extant at the time of the Cornelius (1982) article. The status for Boquita is unknown.
[3]The population may be decreasing in more recent years (Valverde *et al.* 2012).
[4] Low but increasing (Oman 2013).
[5]Extapolated estimate of annual females nesting from nests/km/d (Abreu-Grobois and Plotkin 2008).

B.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 five listing factors (see below). In considering whether a species reclassification or delisting is warranted, we look at each factor singularly and in aggregate and whether these factors contribute to the extinction risk of the species. 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.

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

There are increasing impacts to the nesting and marine environment that affect olive ridley turtles. Structural impacts to nesting habitat include the construction of buildings and pilings, beach armoring and renourishment, and sand extraction (Lutcavage *et al*. 1997, Bouchard *et al*. 1998). 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). These activities have increased in many parts of the olive ridley's range and pose threats to major nesting sites in India and Central America (Cornelius *et al*. 2007). In addition, coastal development is usually accompanied by artificial lighting. The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Witherington and Bjorndal 1991). In many countries, coastal development and artificial lighting are responsible for substantial hatchling mortality. For example, artificial lights increased significantly between 1993 and 2010 in northern Queensland, Australia, exposing nesting olive ridleys to light pollution and potentially disrupting their nesting behavior and disorienting hatchlings (Kamrowski *et al.* 2014).

India's Odisha (formerly known as Orissa) coast hosts one of the largest olive ridley nesting populations (Gahirmatha) and also supports 37–44% of the world's human population as of 1994 (Cohen *et al.* 1997 cited in Mohanty *et al.* 2008). As such, tremendous pressure exists to develop the coast for economic growth, which has led to irreversible impacts to the environment (Mohanty *et. al.* 2008). Sources of artificial lights (e.g., townships, chloro-alkali facilities) misdirected over 90% of the hatchlings from 45 nests in 2004–2005 at Rushikulya, India (Tripathy and Rajasekhar 2009), and light pollution at Rushikulya is predicted to misorient 50% of the hatchlings in any given year (Karnad *et al.* 2009*).* Fewer hatchlings were misoriented in areas of the *Casuarina* (an introduced plant) plantations, but hatchling and egg predation is also higher near the plantations (Muralidharan *et al.* 2013).

Dharma Port, located north of Gahirmatha Marine Sanctuary, India, became operational in 2010. Although the developers sought advice from scientists on how to minimize environmental impacts, concerns remain that maintenance dredging and industrial pollution may adversely impact olive ridleys. Turtle deflector devices were used on dredges and screens were placed over inflow pipes to prevent turtle entrapment. A monitoring program was implemented to document incidental capture of turtles. The developers are working with government officials to propose a lighting ordinance to reduce impacts to sea turtles (see 2008 Marine Turtle Newsletter Issue Number 121 Special Theme Section: Dhamra Port Development, Orissa, India).

At sea there are numerous potential threats including marine pollution, oil and gas exploration, lost and discarded fishing gear, changes in prey abundance and distribution due to commercial fishing, habitat alteration and destruction caused by fishing gear and practices, agricultural runoff, and sewage and industrial discharge (Bramha *et al.* 2011; Lutcavage *et al*. 1997, Frazier *et al*. 2007). There are no data to determine the impacts of these activities to olive ridley populations.

As discussed earlier (see Section A.2.3.2.1), habitat impacts from climate change, especially due to global warming, are likely to become more apparent in future years (IPCC) 2007a) and will impact the ecosystems that sea turtles depend upon (e.g., Doney *et al.* 2012). 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). 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. Dewald and Pike (2013) examined hurricane paths in the northwest Atlantic and northeast Pacific from 1970 through 2007 to quantify the frequency of impacts on sea turtle nesting sites. Approximately 97% of nesting sites (all sea turtle species) in the eastern Pacific and western Atlantic were affected by hurricanes during this period. However, olive ridley nesting beaches were generally in areas exposed to only a few hurricanes over the last 38 years (Dewald and Pike 2013).

Although empirical data on the impacts of destruction, modification and curtailment of the olive ridley's habitat or range are lacking from many areas, habitat loss continues and will likely increase given human encroachment on coastal habitats. Coastal construction, pollution, and other human-related impacts to the olive ridley's habitat will likely increase as coastal human populations expand.

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

Olive ridleys and their eggs have been overutilized worldwide. The history of use and detailed accounts of this use is reviewed by Campbell (2007a), Cornelius *et al*. (2007), and Frazier *et al*. (2007). Use is summarized below by region, with information provided on historical use and contemporary use. There are many "scales" of use and the following summary distinguishes commercial use (of all sizes) from personal use. "Personal use" in this report is meant to imply non-commercial use by individuals or families and includes subsistence use as well as non-subsistence use.

The current impact of human use of olive ridley turtles and their eggs on populations is difficult to evaluate because there are many factors that contribute to a population's growth and decline (e.g., incidental take in commercial fisheries); however, Cornelius *et al*. (2007) identify several solitary nesting beaches and arribada beaches where current egg use is causing declines. Recreational, scientific, or educational overutilization has not been reported for olive ridleys.

In Central and South America, olive ridley eggs have been and still are used for personal and commercial use (Arauz 2000; Campbell 2007a; Cornelius *et al*. 2007; Lagueux 1989). Laws regulating turtle egg use vary among the countries and even where laws prohibit egg use, illegal use of olive ridley eggs is believed to be widespread because enforcement is either non-existent or insufficient. Personal use of turtle eggs is prevalent throughout the region and is viewed as overutilization in some areas, while in other areas it is not viewed as such (Campbell 2007a). The current impact of personal use of eggs on olive ridley abundance and trends in this region is largely unknown; however, on unprotected solitary nesting beaches (most are unprotected), where use often approaches 100%, declines are expected if such use continues.

In Guatemala, eggs are donated to hatcheries run with low budgets and minimal staff. Often hatchlings are retained, sometimes up to 6 days, within the facilities to accommodate 'hatchling races' held on Saturdays at sunset (Handy and Lucas 2010). Retention of hatchlings can raise plasma glucose levels indicating stress (Zenteno *et al.* 2008). These poor hatchery practices likely impact the hatchlings' ability to disperse offshore by depleting energy reserves and affecting the imprinting and navigation processes (Handy and Lucas 2010).

In Nicaragua, commercial egg use occurred in Nicaragua and reportedly led to the disappearance of arribadas at Masachapa and Pochomil in the 1970s (Nietschmann 1975). Egg use still occurs in Nicaragua. Egg collection is prohibited from October 1 to January 31 and year round in protected areas (Valle 1997). Enforcement of this closed period is reportedly poor and very few eggs are left to incubate anywhere in the country (Camacho and Cáceres 1995). Residents collected over 600,000 eggs annually between 1993 and 1999. Egg collection quotas appear to be based on demands of surrounding coastal communities rather than conservation needs of the turtles, and results in chaotic illegal egg commerce (Hope 2002). In 2001, over 100 olive ridleys were documented stranded in the Chacocente Wildlife Refuge, and of the turtles examined (12), 100% (all females) had been cut in the groin area, a common practice by fishermen searching for eggs (Arauz 2000). A subsample of 544 nests at Playa Chacocente and Playa La For monitored during the 2009 and 2011 nesting seasons, showed almost 15% of the nests at Playa Chacocente were poached (Salazar *et al.* 2013).

In Panama, commercial egg use reportedly also occurs (Cornelius *et al*. 2007); however, the extent of the use and its impact on the nesting population is undocumented.

In Costa Rica, the largest commercial egg use occurs in Ostional, where a regulated collection of olive ridley turtle eggs supplies a national market. This use was largely unregulated 40 years ago but has been legal and regulated to varying degrees since 1987 (Campbell 2007b; Madrigal-Ballestero *et al*. 2013). Local citizens legally harvest eggs through a local organization that establishes harvest levels and monitors compliance. The organization's income from the harvest has been approximately $400,000 (U.S. $) annually since 2010; of which, 70% of the total income is equally distributed to individual harvesters and 30% goes to administrative costs and local infrastructure (e.g., roads, bridges) projects (Madrigal-Ballestero *et al*. 2013). Compliance with the harvest levels appears to be affected by an individual harvester's reliance on the income from the sale of eggs and their perception of the legitimacy of the rules governing the harvest (Madrigal-Ballestero *et al*. 2013). Historically, the percentage of eggs harvested for the commercial market ranged from 5.4% to 38.6% annually (1988-1997), depending on the season (Ballestero *et al*. 2000). From 2006-2010, overall harvest averaged about 21% per year (Valverde *et al.* 2012). Although rare, 100% harvest has been documented in some months and legal harvest should be closely monitored to ensure its sustainability (Valverde *et al.* 2012). Poaching is still a concern.

In the western Atlantic, olive ridleys were also overutilized (Cliffton *et al*. 1982, Green and Ortiz-Crespo 1982, Campbell 2007a). Both casual and organized take of adults and eggs of all nesting sea turtle species historically were widespread in the Guianas and northeast Brazil.

In Suriname, about 1,500 nesting olive ridleys were killed annually during most of the 1930s (Geijskes 1945 as cited in Reichart and Fretey 1993). The direct take of adults apparently diminished over time, but egg collection was intense and reached nearly 100% in the late 1960s (Schulz 1975). Despite a Suriname law that banned egg use in 1970, uncontrolled egg collection occurred from the late 1980s to the early 1990s at Eilanti Beach and elsewhere (Reichart 1993, Reichart and Fretey 1993). Illegal use is still believed to be widespread. Hoekert *et al*. (1996) reported that more than 40% of olive ridley nests were collected during the peak season in 1995.

In Brazil, initial surveys of sea turtle nesting activity in the early 1980s revealed unorganized but widespread use of adults and eggs of all species nesting along the Sergipe coast (Marcovaldi and Marcovaldi 1999).

In the eastern Atlantic Ocean, olive ridleys and their eggs are used along the entire coast of West Africa and sold in local and regional markets. During the 1992 through 1994 nesting seasons in Orango National Park, Guinea-Bissau, 26% (37 of 142) of the nests were predated by humans (Catry *et al.* 2009). Surveys of fishing communities in Angola indicate widespread adult and nest harvest for subsistence use during 2005–2006, including 100% harvest of females and eggs on some beaches (Weir *et al.* 2007). A survey of 27 West African countries (including Macaronesia) indicated that nesting females were killed in 14 of them (Fretey 2001). The extent of use and its impact on populations in the region is undocumented.

In the Indian Ocean, use of adult olive ridleys and their eggs for personal and commercial use has been widespread (Frazier 1982; Frazier *et al*. 2007). Use of turtle eggs for human consumption and domestic animal consumption historically was widespread in the Indian Ocean and continues today largely wherever ridleys nest (Cornelius *et al*. 2007). Commercial use of olive ridley eggs once occurred at the arribada beach in Gahirmatha, India, and in Myanmar and resulted in the collection of hundreds of thousands of eggs annually (Cornelius *et al*. 2007).

Egg use has been reported in India, Andaman Islands (Union Territory of India), Bangladesh, Myanmar, Sri Lanka, Pakistan, and Malaysia and is believed to have caused the decline of olive ridleys in these countries (Cornelius *et al*. 2007). Personal subsistence use of adult olive ridley turtles is also fairly widespread (Cornelius *et al*. 2007; Frazier *et al*. 2007). In Sri Lanka at least seven hatcheries operate under private ownership, of which many were damaged by the 2004 tsunami (Rajakaruna *et al.* 2013). All hatcheries purchase eggs from fishermen or villagers, and one hatchery owner reported that police donate eggs confiscated from poachers. Eggs are often collected in plastic bags rather than the recommended polystyrene boxes and hatchery owners are unaware of when the eggs were laid. Incubation conditions are poor, likely affecting sand temperatures and skewing sex ratios. Hatchlings were held in tanks for 24 or more hours, sometimes for days, where the hatchlings swam continuously in the tank and the yolk sac was completely absorbed (Rajakaruna *et al.* 2013).

In-water harvest has severely impacted olive ridley populations. As discussed earlier (see Section A.2.3.2.2), olive ridleys were overutilized for commercial purposes in two legal turtle fisheries that operated in the eastern Pacific Ocean (Campbell 2007a; Cliffton *et al*. 1982; Green and Ortiz-Crespo 1982), which may have affected the Threatened populations. The Mexican turtle fishery caused rapid, large declines at olive ridley arribada beaches in Mexico (Cliffton *et al*. 1982) that were so dramatic they have been widely referred to in the literature as population collapses, crashes, or extinctions. An estimated 2 million turtles were taken for their meat and leather until 1990 when the fishery closed (Aridjis 1990). The closure of the olive ridley turtle fishery has decreased the threat to the population. However, illegal take of adult turtles still occurs in the region and the impact of this take is unknown. There is evidence that thousands of olive ridleys are still taken each year along the Pacific coast of Mexico (Frazier *et al*. 2007). The

Mexican enforcement agency, Procuraduria Federal de Protección al Ambiente (PROFEPA), seized approximately 1,000-8,000 kg of turtle meat, 100–1,800 units of turtle leather, and several hundred dead and live whole turtles each year in the State of Oaxaca (species not specified) (Trinidad and Wilson 2000).

An Ecuadorian turtle fishery also existed during the 1970s and fished several hundreds of thousands of olive ridleys during this time (Green and Ortiz-Crespo 1982). In 1978 alone, 80,535 to 89,483 turtles were harvested (Green and Ortiz-Crespo 1982). This fishery is also believed to have contributed to the decline in the number of olive ridleys nesting on Mexican arribada beaches. A direct link between Mexico nesting beaches and Ecuadorian waters was established when olive ridleys, tagged while nesting in Mexico, were later captured in the Ecuadorian turtle fishery (Green and Ortiz-Crespo 1982).

In summary, the harvest of nesting turtles and eggs and illegal take in fisheries continues to be widespread and poses a significant threat to the Threatened populations.

B.2.3.2.3 Disease or predation:

Little is known about disease in olive ridleys (George 1997). Nothing is known about the impact of disease on olive ridley abundance. The only disease identified in the literature thus far for olive ridleys is fibropapillomatosis, sometimes associated with a herpes-virus found in sea turtles nearly worldwide (Herbst 1994). The incidence of fibropapillomatosis is not believed to be high in olive ridleys. However, the disease has been observed in olive ridleys nesting in Costa Rica (Aguirre *et al.* 1999; Herbst 1994), and India (Kartik Shanker, Indian Institute of Science, personal communication).

Over 1,000 turtles, of which 99% were olive ridleys, stranded dead within a two-month period on the coast of Ecuador in 1999 (Alava *et al.* 2005). The causes of the strandings are unknown; however, Alava *et al.* (2005) cite epizootic outbreaks as one possibility.

Predation on olive ridleys, their eggs, and offspring occurs on land and in the ocean throughout their range, and the relative impacts of this mortality on nesting populations is unknown. Eggs and hatchlings fall prey to numerous mammalian, avian, reptilian, invertebrate, and fungal organisms, including wood storks (Burger and Gochfeld 2013), beetles (Harfush *et al.* 2008b), white-nosed coatis, jackals, hyenas, feral dogs, and pigs (Barquero-Edge 2013; Cornelius and Robinson 1982, Eckrich and Owens 1995; Tripathy and Rajasekhar 2009). Solitary nesting beaches in India in the nesting seasons 2000-2001 and 2004-5 reported 52-68% predation of the nests, mostly by animals (Wesley Sunderraj 2012). Over two nesting seasons from 2003-2005, predation at Rushikulya was as high as 83% for sporadic nests, but low for arribada nests (about 2-8%) (Tripathy and Rajasekhar 2009). On four nesting beaches on the Osa peninsula, Costa Rica, 1,300 nests were recorded in 2010, of which 67% were preyed upon, including harvest of eggs (Barquero-Edge 2013). At Playon de El Valle, Colombia, nest predation is a major threat and can be as high as 100% in some years (Barrientos-Muñoz and Ramirez-Gallego 2012).

On land, adult females fall prey to crocodiles (Ortiz *et al*. 1997; Whiting and Whiting 2011; Whiting *et al.* 2007b), coyotes (Cornelius and Robinson 1982; P. Plotkin, Cornell University, personal observation), and jaguars (Cornelius and Robinson 1982; Kelle *et al.* 2004). In the ocean, sharks, billfish, and whales may prey on hatchlings and adult turtles (Frazier *et al*. 1994, 1995; Pitman and Dutton 2004). The gut contents of a mahi mahi caught south of Mazatlan, Mexico, contained hatchlings in early digestive state indicating the hatchlings were likely eaten just off nesting beaches in the southern tip of Mazatlan (Villasñor *et al.* 2010).

In summary, disease and predation are believed to be relatively minor threats to the Threatened populations. The best available data suggest that current nest and hatchling predation on several nesting beaches and in water habitats is a potential threat but does not indicate severity or whether the population trend is affected by the predation rate.

B.2.3.2.4 Inadequacy of existing regulatory mechanisms:

The migratory nature of olive ridleys requires international collaboration to ensure their survival. For the purposes of the Threatened populations section, we consider instruments (e.g., regulations, treaties, conventions, agreements) that relate to the conservation and recovery of olive ridleys globally. A summary of the main global instruments (e.g., regulations, treaties, conventions, agreements) that relate to the conservation and recovery of olive ridleys is provided below.

In 2009, the United States established the Mariana Trench, Rose Atoll, and Pacific Remote Islands Marine National Monuments, which prohibited commercial and recreational fisheries in an area encompassing over 95,000 square miles. Under the MSA (described in section A.2.3.2.4 and below), 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. Olive ridley 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; 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. Since 2001, a large time and area closure for the California-based large mesh drift gillnet fishery targeting swordfish/common thresher shark off the U.S. west coast has significantly reduced leatherback sea turtle interactions and may have benefited olive ridleys.

Conservation programs geared to protect olive ridley nests are ongoing in many areas and are too numerous to mention. For example, the Trust for Environment Education Foundation started a volunteer monitoring program in 2002 to protect nests along the coast of Chennai, India (Dharini 2008). Over 400 volunteers from local fishing villages joined the efforts, and in 2006 no illegal poaching or nests depredation was reported. In Myanmar, a sea turtle conservation and management program has been ongoing since 1963. From 2001 to 2008, the Myanmar Department of Fisheries recorded 360 nests and 35,709 hatchlings released from Gadongalay Island (Nwe and Lwin 2013).

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 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-Solomon Seas Ecoregion: Tri-National Turtle Agreement
In 2006, Indonesia, Papua New Guinea, and the Solomon Islands signed the Tri-National Turtle Agreement to protect olive ridleys. 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 January 2014. 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. In 1999, the parties entered into a Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa. It aims to safeguard six marine turtle species - including the olive ridley - 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.

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 180 Parties as of February 2014. The most recent Parties are the Angola and Iraq who signed in 2013 and 2014, respectively. 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 olive ridley is listed on Appendix 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 olive ridleys 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 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.
As mentioned previously, in 2007, the IATTC passed as resolution (C-07-03) that encompasses most of the elements contained in the WCPFC's resolution passed one year later, including the require to use dipnets, line cutters, etc. when handling sea turtles; however, the IATTC resolution only requires parties to expeditiously undertake research to test the use of mitigation measures required elsewhere to reduce sea turtle bycatch in longlines (e.g., circle hooks and finfish bait). The IATTC has also developed a memorandum of understanding with the Inter-American Convention for the Protection and Conservation of Sea Turtles. The International Commission for the Conservation of Atlantic Tunas (http://www.iccat.int) has a recommendation on sea turtles, which calls for implementing the FAO Guidelines for sea turtles, avoiding encirclement of sea turtles by purse seiners, safely handling and releasing sea turtles, and reporting on interactions. The Commission does not have any specific gear requirements in longline fisheries. The International Commission for the Conservation of Atlantic Tunas is currently undertaking an ecological risk assessment to better understand the impact of its fisheries on sea turtle populations. The Indian Ocean Tuna Commission (http://www.iotc.org/) is also in the process of carrying out an ecological risk assessment for sea turtles. Their turtle measures encompass similar elements of the other organizations but do not require the use of certain gear or bait in longline fisheries. Finally, the Commission for the Conservation of Southern Bluefin Tuna

(http://www.ccsbt.org) supports the measures called for in the Western and Central Pacific Fisheries Commission and the Indian Ocean Tuna Commission (http://www.wcpfc.int/node/591).

Other international fisheries organizations that may influence olive ridley 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 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).

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.

In 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. 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).

Discussed earlier (see Section A.2.3.2.4), the Services concluded the effectiveness of domestic and intergovernmental authorities (e.g., closure of the turtle fishery and prohibition of egg harvest) were effective for the Endangered breeding colony populations in Mexico as evidenced by the in-water abundance estimates and the increases in nests reported for most nesting beaches. However for the Threatened populations, the effectiveness of domestic and intergovernmental authorities, conservation and protection measures is inconsistent throughout the species' global distribution and many populations continue to decline. For these reasons and notwithstanding the growing number of domestic and intergovernmental authorities, the Services believe that the Threatened populations of olive ridleys remain threatened because of the inadequacy of existing regulatory mechanisms for their protection. However, we have substantial 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 olive ridley. Depending on the outcome of that analysis, some Threatened populations may warrant reclassification to Endangered status. See Section 4.0, for additional information.

B.2.3.2.5 Other natural or manmade factors affecting its continued existence:

Several manmade factors affect olive ridleys in foraging areas and on nesting beaches. Two of these are truly global phenomena: climate change and fisheries bycatch. As stated earlier (Section B.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). Climate change will impact the ecosystems that sea turtles depend upon (e.g., Doney *et al.* 2012). 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 olive ridleys exhibit temperature-dependent sex determination (reviewed by Wibbels 2003, 2007), there may be a skewing of future cohorts toward a strong female bias since warmer temperatures produce more female embryos (Hawkes *et al*. 2009; Sifuentes-Romero *et al*. 2013). More importantly, elevated sand temperatures can result in almost zero hatch success (Sifuentes-Romero *et al*. 2013). The effects of global warming are difficult to predict, but changes in reproductive behavior (e.g., remigration intervals, timing and length of nesting season) may occur (reviewed by Hamann *et al.* 2013; Hawkes *et al.* 2009). 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).

Cyclones are common along the Odisha (formerly known as Orissa) coast and result in destruction of vegetation and severe erosion of the coastline (Mohanty *et al*. 2008). One of the largest rookeries, Gahiramatha beach, India, was 25 km long in the 1980s, but due to cyclones, the beach has eroded down to only 1.5 km (Senapati 2013). Nesting habitat loss has significantly affected hatch success at Gahirmatha (20.4% ± 17.9%) compared to Rushikulya, (89.5% ± 10.3) where the rate of erosion is lower (Behera *et al.* 2013). Signficant beach erosion resulted in a loss of 60% of the nests in the 2003-2004 season at Rushikulya on the Odisha (formerly known as Orissa) coast (Tripathy and Rajasekhar 2009).

Programs to restore the beach include planting native vegetation (Kabi 2013), but whether these efforts successfully preserve nesting beach habitat has not been fully evaluated.

In 2004, a major earthquake occurred off the west coast of Sumatra, Indonesia, resulting in tsunami waves measuring 30 meters high affecting large parts of the Pacific and Indian Oceans. The 2004 tsunami struck many nesting beaches leaving consequences for later nesting season success. Post-tsunami assessments claimed that vegetation had protected life and property. As a result, a campaign to plant *Casuarina equisetifolia* trees was implemented to prevent further erosion and protect human property. The trees hinder females who come to shore to nest (Velusamy and Sundararaju 2009), increase predation of nests by providing habitat for predators, and increase shade which can alter sex ratios (Chaudhari *et al.* 2009; Tripathy and Rajasekhar 2009).

The incidental capture of olive ridleys that occurs worldwide in fisheries is a major concern (e.g., Wallace *et al.* 2010a, 2013) and occurs in trawl fisheries, longline fisheries, purse seines, gillnet and other net fisheries, and hook and line fisheries (Frazier *et al*. 2007). The impact of the incidental capture of olive ridleys in fisheries has been well documented for some regions but not for others. In some locations where bycatch statistics are unavailable from fisheries, cause and effect has been used to implicate a fishery in the decline of olive ridleys.

In the eastern Pacific Ocean, olive ridleys were the second most common species caught from 1999 to 2010 in the Costa Rican longline fishery (Dapp *et al.* 2013). Observer coverage varied between years and seasons, but total bycatch was 2,864, of which 277 were adult females and 362 adult males. The mean catch rate was 8.85 olive ridleys per 1,000 hooks (Dapp *et al.* 2013).

In the United States, olive ridley bycatch was only reported for fisheries operating in the Pacific Ocean (Finkbeiner *et al.* 2011). Measures to reduce or minimize the effects of the take in these fisheries appear to be successful (see Section A2.3.2.5.).

In the central and western Pacific Ocean, several thousand longline vessels operate, representing over 20 countries, although some of the smaller Pacific island countries have relatively few vessels. Taiwan's offshore fleet is relatively large, with approximately 1,600 vessels on average based on data through 2005 (Fahy 2011). From available data up until the late 1990s, longline fisheries in this area were estimated to take over 2,000 sea turtles per year, with 500-600 expected to die (23-27% mortality rate). The majority of the sea turtle species taken were olive ridleys (Oceanic Fisheries Programme, Secretariat of the Pacific Community 2001). Japanese tuna longliners are known to interact with sea turtles. Data from 2000 indicate approximately 6,000 turtles are caught annually, with approximately 50% mortality (K. Hanafusa, Fisheries Agency of Japan, personal communication, 2004). Species composition is unknown, but interactions with olive ridleys are likely. Coastal gillnets in Taiwan are documented to interact with sea turtles. According to interviews with fishermen, 14 olive ridleys were taken in the fishery from 1991-1995 (Cheng and Chen 1997). Prior to the mandatory use of TEDs in 2000, a large number of sea turtles (between 5,000 and 6,000 per year) were taken in the Australian fisheries shrimp trawl fisheries, with a mortality rate of around 40%. Following the use of TEDs, the number of turtles has been reduced to below 200 per year (Robins *et al*. 2002). In general, olive ridleys were the second most-commonly caught species in these fisheries.

In the northwestern Sulu Sea off the Philippines, incidental capture in local fisheries are a continuing threat (Bagarinao 2011).

In the western Atlantic Ocean, olive ridleys are caught in shrimp trawl fisheries, specifically along the Guianas and Suriname coasts. This bycatch is believed to be the main cause of the significant population decline observed there since the 1970s. The number of olive ridleys captured incidentally in trawl fisheries off the coasts of Suriname and French Guiana is believed to be approximately several thousand turtles annually (Frazier *et al*. 2007; Godfrey and Chevalier 2004; Tambiah 1994). Continued mortality from shrimp trawling appears to be the major threat to the recovery of these nesting populations (Frazier *et al*. 2007; Godfrey and Chevalier 2004). Gillnets and other fishing methods in this region also capture olive ridleys incidentally but to a lesser extent than shrimp trawl fisheries (Frazier *et al*. 2007). Shrimp trawling off the nesting beaches of Sergipe, Brazil, are a major source of mortality for nesting olive ridleys (da Silva *et al.* 2010). By 2003, three management measures were in place to reduce sea turtle bycatch in the shrimp trawl fishery: (1) fishing is prohibited within 3 nm of the Sergipe coast; (2) fishing is closed from May through mid-June each year; and (3) TEDs are mandatory. However, lack of compliance and enforcement have resulted in poor results as evidenced by high sea turtle strandings in the area (da Silva *et al.* 2010).

In the eastern Atlantic, the incidental capture of olive ridleys by commercial fisheries is thought to be a significant threat; however, there is very little systematic data on incidental capture of marine turtles in West Africa (Frazier *et al*. 2007). Olive ridleys have been observed entangled in discarded fishing gear in waters off Angola (Weir *et al.* 2007). In the Republic of Congo, a community-based program, including fishers, implemented a release program for turtles captured incidental to fishing operations. From 2005 through 2010, the program responded to 1,551 olive ridleys caught in fishing gear, of which 1,270 were treated and released (Girard and Breheret 2013). Taiwanese longline vessels targeting albacore and bigeye tuna operating in the Atlantic Ocean caught 175 olive ridleys on 103 trips and 13,096 observed sets from 2004 to 2011 (Huang 2013). Olive ridley size ranged from 40–70 cm straight carapace length. Overall bycatch rate (all sea turtle species combined) was zero/1000 hooks in the north Atlantic Ocean from October to December and 0.0311 sea turtles/1000 hooks in the tropical areas of the Atlantic Ocean from April to June (Huang 2013). In 2012, olive ridleys were the most common species bycaught in artisanal gillnet fisheries operating off the coast of Ghana with an average catch of 2.96 turtles per boat over 34 days (Tanner 2014).

In the Indian Ocean, incidental capture of olive ridleys is extremely high along the coast of Odisha (formerly known as Orissa), India, where the densest concentrations of olive ridleys gather to nest and fishing effort is high. Trawling effort increased four-fold from 1980 to 2005 in the offshore waters of Odisha and was positively correlated with sea turtle strandings in 2008/2009 (r = 0.80) and 2009/2010 (r = 0.91) (Subrata *et al.* 2013). A total of 14,035 turtles were counted stranded during 2008/2009 and 3,481 in 2009/2010 (Subrata *et al.* 2013). In the 1990s, recorded carcasses increased from 5,000 in 1994 to 15,000 in 1999, although it is unknown whether monitoring effort varied over time (Pandav and Choudhury 1999). Between 1996–2001 approximately 75,000 dead turtles were counted on the Odisha coast (Wright and Mohanty 2002). Although TEDs are mandatory off Odisha and there are prohibitions on mechanized fishing within 5 km off the coast, the regulations are likely not frequently enforced or followed (Shanker *et al*. 2003b). A gillnet fishery also operates in the region and contributes to the ridley mortality observed along this coastline. In 2001, a gillnet washed ashore near Gahirmatha with over 200 dead turtles entangled in it and over 10,000 turtles stranded that year (Wright and Mohanty 2002), indicating a serious threat from this fishery. In Eritrea, turtle excluder devices are required in all trawl fisheries and fishing is prohibited in shallow waters off the islands and mainland to protect sea turtles (Mebrahtu 2013).

Ghost nets may pose a serious threat to juveniles in waters off the Maldives. Anecdotal reports of olive ridley juveniles entangled in fishing gear showed 34 of 45 records were of entanglement in pieces of lost fishing nets (Anderson *et al.* 2009). In the Gulf of Carpentaria, up to 3 tons/km of derelict fishing gear have been collected in cleanup operations (Wilcox *et al.* 2012). Sea turtles, including olive ridleys, become entangled in the gear and are injured or die. Genetic analysis of olive ridleys entangled in these ghost nets indicate turtles come from nesting populations within the Northern Territory, but also haplotypes not found in the Northern Territory were recorded. Thus, these ghost nets are likely impacting nesting populations over a large geographical area (Jensen *et al.* 2013). Most of the derelict gear enters the Gulf of Carpentaria from the northwest and moves along the northeastern shore in a clockwise pattern. Aerial or satellite monitoring of the area where gear enters the Gulf of Carpentaria would allow for intercepting the gear before it disperses along the coast, killing wildlife in its wake (Wilcox *et al.* 2012).

Other risk factors for sea turtles include interbreeding. Olive ridley females have been documented to interbreed with male loggerhead (*Caretta caretta*) sea turtles in Sergipe, Brazil (Reis *et al.* 2010), but the Services do not believe hybridization presents a global threat to the recovery of the Threatened populations.

Increased exposure to heavy metals and other contaminants in the marine environment also affect olive ridleys (Bonzi *et al.* 2013). 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).

In summary, incidental capture in commercial and subsistence fisheries remain a serious threat to the recovery of the Threatened populations. The emerging threat of impacts from climate change resulting in skewed sex ratios, embryo mortality, and loss of habitat due to severe storms and sea level rise are likely to increase in the foreseeable future.

2.4 Synthesis

Endangered Populations (Mexico breeding populations)

The current abundance of olive ridleys compared with historical abundance at each of the large arribada beaches indicates the populations experienced steep declines due to over-exploitation. The only exception may be Ixtapilla, which was not discovered until 1994 and long-term nesting trends are unknown. Based on the current number of olive ridleys nesting in Mexico, three populations appear to be stable (Mismaloya, Tlacoyunque, and Moro Ayuta), two increasing (Ixtapilla, La Escobilla) and one decreasing (Chacahua). Nesting trends in Mexico, where known, at non-arribada beaches are stable or increasing in recent years. The trend data are generally less than the 10-year period specified in delisting recovery criterion no. 3 in the recovery plan (see Section 2.2.1). Recent at-sea estimates of density and abundance of the olive ridley show a yearly estimate of 1.39 million (Confidence Interval: 1.15 to 1.62 million), which is consistent with the increases seen on the eastern Pacific nesting beaches as a result of protection programs that began in the 1990s. The closure of the olive ridley turtle fishery and ban on egg harvest has decreased the threat to the population. Although illegal harvest continues, the Endangered populations appear to have stabilized from the previous population collapse due to over exploitation.

Threatened Populations (globally except Mexico breeding populations)

In the eastern Pacific, the large arribada nesting populations have declined since the 1970s. Nesting at some arribada beaches continues to decline (e.g., Nancite in Costa Rica) and is stable or increasing at others (e.g., Ostional in Costa Rica). There are too few data available from solitary nesting beaches to confirm the declining trend that has been described for numerous countries throughout the region including El Salvador, Guatemala, Costa Rica, and Panama.

Western Atlantic arribada nesting populations are currently very small. Data indicate the Suriname/French Guiana nesting population may still be threatened by incidental capture in the shrimp trawl fishery. The Suriname olive ridley population is currently small and has declined by more than 90% since the late 1960s. However, nesting is reported to be increasing in French Guiana. The other nesting population in Brazil, for which no long term data are available, is small, but increasing. In the eastern Atlantic, long-term data are not available and thus the abundance and trends of this population cannot be assessed at this time. However, the threats associated with growing commercial and artisanal (i.e., generally smaller scale local, non-commercial) fisheries in the region are serious and warrant close attention.

In the northern Indian Ocean, arribada nesting populations are still large, but trend data are ambiguous and major threats continue. Development of nesting beaches and high levels of fisheries bycatch from shrimp trawl and gillnets continues off nesting beaches, along migratory routes and on foraging grounds are a concern. Declines of solitary nesting olive ridleys have been reported in Bangladesh, Myanmar, Malaysia, Pakistan, and southwest India.

3.0 RESULTS

3.1 Recommended Classification:

3.1.1 Endangered populations

Based on the best available information, we conclude the breeding colony populations on the Pacific coast of Mexico may warrant reclassification. The Services have based this on the increasing trend in-water and on the nesting beaches as a result of protection on the beaches, reduced threats in the water as a result of elimination of the directed fishery, and the effectiveness of domestic and intergovernmental regulations.

However, for the current population listings for the olive ridley (both Endangered and Threatened), 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 olive ridley. See Section 4.0 for additional information. Thus, the Services recommend that the global status review be completed prior to any reclassification of existing olive ridley listings.

3.1.2 Threatened populations

Based on the best available information, we conclude the threatened olive ridley populations should not be delisted. 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 olive ridley after which time status of DPSs can be determined. Pending the oucome of the DPS policy, if some populations are determined to be a DPS, they may warrant reclassification to Endangered status. The Services have based this on the decreasing trend in the large arribada nesting populations, continued threats in-water and on the nesting beaches, and the inconsistent effectiveness of domestic and intergovernmental regulations and enforcement throughout the species' global distribution. See Section 4.0, for additional information.

3.2 New Recovery Priority Number: No change.

4.0 RECOMMENDATIONS FOR FUTURE ACTIONS

We have substantial 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 olive ridley. 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 begun to assemble and analyze this new information, which indicates, at a minimum, a separation of populations by ocean basins and likely substructuring within Pacific and Indian Ocean basins. To determine the application of the DPS Policy to the olive ridley, the Services intend to fully assemble and analyze this new information in accordance with the DPS Policy. See Section A.2.3 and B.2.3 for new information since the last 5-year review.

The current Recovery Plan for U.S. Pacific Populations of the Olive Ridley was completed in 1998. The recovery criteria contained in the Plan, while not strictly adhering to all elements of the 2006 NMFS Interim Recovery Planning Guidance, are a viable measure of the species status. The recovery plan needs to identify and incorporate recovery criteria for both the Endangered (reclassification to Threatened and delisting) and Threatened (delisting) populations. The species biology and population status information can be updated. While some additional recovery actions can be identified, the Services conclude that the current Plan remains a valid conservation planning tool. The Recovery Plan should be re-examined over the next 5-10 year horizon, particularly if the DPS analysis results in restructuring of the current listing, to update the plan to conform to the Services Interim Recovery Planning Guidance. In the near-term, additional information and data are particularly needed on genetic relationships among nesting populations, impacts of fisheries (particularly trawl and longline fisheries) on population status, foraging areas and identification of threats at foraging areas, and long-term population trends.

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

Current Classification: Endangered and Threatened

Endangered Population - breeding colony populations on Pacific coast of Mexico
Threatened Populations - wherever found except where listed as Endangered

Recommendation resulting from the 5-Year Review: Reclassification of the Endangered Population to Threatened may be warranted. No change to Threatened Populations.

Review Conducted By:
Therese Conant, Angela Somma (National Marine Fisheries Service)
Ann Marie Lauritsen, Kelly Bibb, Earl Possardt (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: MAY 30 2014
Donna S. Wieting

Assistant Administrator, NOAA Fisheries

✓ Concur ____ Do Not Concur

Signature ____________ Date 6/3/14

U.S. FISH AND WILDLIFE SERVICE
5-YEAR REVIEW of *Olive Ridley Sea Turtle*

Current Classification: Endangered and Threatened

Endangered Population - breeding colony populations on Pacific coast of Mexico
Threatened Populations - wherever found except where listed as Endangered

Recommendation resulting from the 5-Year Review: Reclassification of the Endangered Population to Threatened may be warranted. No change to Threatened Populations.

Review Conducted By:
Therese Conant, Angela Somma (National Marine Fisheries Service)
Ann Marie Lauritsen, Kelly Bibb, Earl Possardt (U.S. Fish and Wildlife Service)

FIELD OFFICE APPROVAL:

Lead Field Supervisor, Fish and Wildlife Service

Approve: ______________________ Date: 6/2/2014
Jay B. Herrington

REGIONAL OFFICE APPROVAL:

Lead Regional Director, Fish and Wildlife Service
Acting

Approve: ______________________ Date: 6/18/14