

Lepidochelys olivacea

Scientific name (including authority details):

Lepidochelys olivacea (Eschscholtz, 1829)

Synonym/s (if there has been a taxonomic change in the last 5 years or if widely used):

None

English Common Names (if known):

Olive ridley, Pacific ridley

Other Common Names (if known and state language):

French: Ridley du Pacifique, tortue *bâtarde*, tortue olivâtre

Spanish: tortuga golfina, tortuga lora, tortuga olivacea, tortuga guaraguá, tortuga parlama, tortuga carpintera, tortuga mulato, tortuga manila

Portuguese: tartaruga-oliva, tartaruga-pequena, tartaruga-comum

Order	Family	Subfamily
Testudines	Cheloniidae	Chelonini

Geographic Range of species (figure 1):

The olive ridley sea turtle has a circumtropical distribution, with nesting occurring throughout tropical waters (except the Gulf of Mexico) and migratory circuits in tropical and some subtropical areas (Atlantic Ocean – eastern central, northeast, northwest, southeast, southwest, western central; Indian Ocean – eastern, western; Pacific Ocean – eastern central, northwest, southwest, western central) (Pritchard, 1969). Nesting occurs in nearly 60 countries worldwide. Migratory movements are less well studied than other marine turtle species but are known to involve coastal waters of over 80 countries (see Table 1). With very few exceptions they are not known to move between ocean basins or to cross from one ocean border to the other. Within a region, olive ridleys may move between the oceanic and neritic zones (Plotkin et al., 1995; Shanker et al., 2003a) or just occupy neritic waters (Pritchard, 1976; Reichart, 1993).

Habitat and Ecology Information:

Habitats. Like most other sea turtles, olive ridleys display a complex life cycle which requires a range of geographically separated localities and multiple habitats (Márquez, 1990). Females lay their nests on coastal sandy beaches from which neonates emerge and enter the marine environment to continue their development. They remain in a pelagic phase, drifting passively with major currents that disperse far from their natal sites, with juveniles sharing some of the adults' habitats (Kopitsky et al., 2000) until sexual maturity is reached (Musick and Limpus, 1997). Reproductively active males and females migrate toward coastal zones and concentrate near nesting beaches. However, some males appear to remain in oceanic waters and mate with females en route to their nesting beaches (Plotkin et al. 1996; Kopitsky et al., 2000). Their post-breeding migrations are complex, with pathways varying annually (Plotkin, 1994) and with no apparent migratory corridors, swimming hundreds or thousands of kilometers over large ocean expanses (Morreale et al., 2007), commonly within the 20°C isotherms (Márquez, 1990). In the East Pacific, they are present from 30°N to 15°S and often seen within 1,200 nautical miles from shore although they have been sighted as far as 140°W (IATTC, 2004). Western Atlantic olive ridleys appear to remain in neritic waters after breeding (Pritchard, 1976; Reichart, 1993).

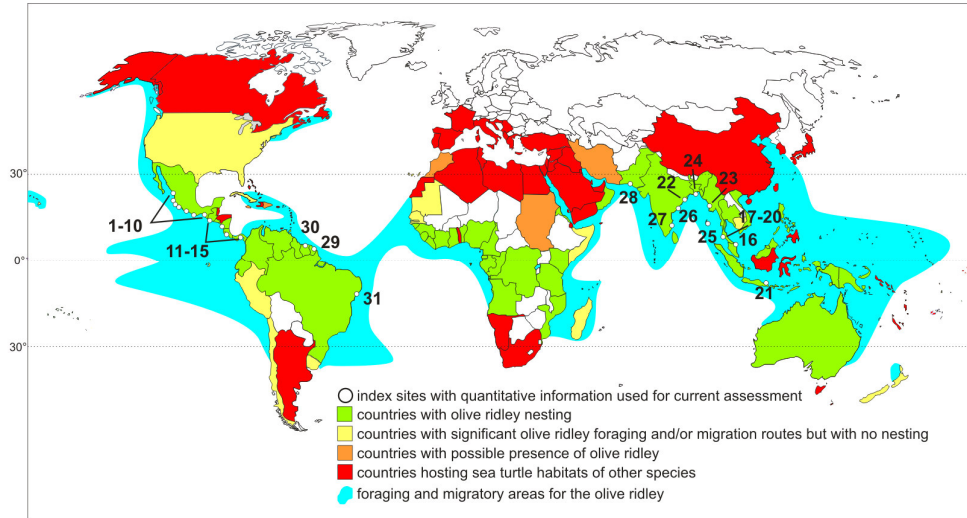


Figure 1. Geographic range of habitats utilized by the olive ridley, *Lepidochelys olivacea*, based on data in Table 6.

Demographic features / Reproduction. The species displays three modes of reproduction: arribada, dispersed nesting, and mixed strategy (Bernardo and Plotkin, 2007). The first mode represents a synchronous, mass nesting behavior that may include hundreds to thousands of females over a period of days and occurs in fewer than a dozen places worldwide. The more common form of nesting is dispersed or “solitary” with no apparent synchronicity between individual events. At some localities, a mixture of these two forms of nesting can also occur. In general, individual olive ridleys may nest one, two or three times per season, with approximately 100-110 eggs per clutch (Pritchard and Plotkin, 1995). For this assessment we have used an average number of 2.5 nests/female/season and 105 eggs/nest. In contrast to other sea turtle species, the reproductive cycle is nearly annual (over 60% of turtles nest every year; Márquez, 1990). Solitary nesters oviposit on 14 day cycles whereas arribada nesters approximately every 28 days (Pritchard, 1969; Kalb and Owens, 1994; Kalb, 1999). Kalb (1999) found that within a nesting season solitary nesters use multiple beaches for oviposition but arribada nesters display nest site fidelity. There are extreme variations in hatching rates between nesting beaches, however, in general they are much higher in solitary nesting beaches where around 80% is common and sometimes even higher (Gaos et al., 2006). It is widely recognized that survivorship is extremely low on high density arribada nesting beaches because of density-dependent mortality (Cornelius et al., 1991) leading to hatching rates as low as 1 to 8% (Cornelius et al., 1992). 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. On solitary nesting beaches, where density-dependent mortality is not a factor, hatching rates are significantly higher (Castro, 1986; Gaos et al., 2006). Post-hatching survivorship is unknown and there is no information available on recruitment rates. Presumably, like other sea turtles, olive ridleys experience high mortality in their early life stages. Juveniles are believed to occur in similar habitats as the adults (i.e. pelagic waters) where they forage on gelatinous prey such as jellyfish, salps and tunicates (Kopitsky et al., 2004).

Generation Length

For the purposes of Red List assessments, generation length is defined as the “average age of parents” (IUCN, 2001). Since this information is not available from direct observation of sea

turtle species we derived a comparable value from estimates of age at which 50% of the breeders are expected to have survived, using information for age at sexual maturity and adult survival rates. An important caveat is that, while it is known that different populations of the same species can attain sexual maturity at different ages (Heppell et al. 2003) and therefore different values would need to be taken into account for different regions, the information is only available for a single region and we have had to assume the estimated values are generally applicable on a global scale. The only published study on growth and age for olive ridleys (Zug et al. 2006) indicates a mean age at sexual maturity for North-central Pacific ridley sea turtles of around 13 years (range of 10-18 years). We calculated the time it would take for a cohort of breeders to reach 50% of its original size from $S^n = 50\%$, where n is years since age at first reproduction, and S is annual survival. Solving for n , $n = \ln(0.5)/\ln(S)$. As extensive estimates of annual survival rates for female nesters are only available from the better studied sister taxon, *Lepidochelys kempii*, (TEWG, 2000) we used these, which have a range of 85-92% per year. Thus, $n_{50\%} = 4 - 9$ yrs and our derived average age of female olive ridley parents is 17-22 years. We additionally assumed that this value would be the same for male parents. For simplicity, we have used a value of 20 yrs for the species' generation length in this assessment.

Major Threats.

Like other long-lived species, olive ridleys are prone to population declines because of slow intrinsic growth rate in combination with anthropogenic impacts. These can accumulate over a protracted development through various life stages, multiple habitats (nesting beaches, migratory routes and pelagic foraging zones) and vast geographic expanses.

Targeted exploitation.

Egg harvest. Olive ridleys and their eggs have been harvested, mostly unsustainably, worldwide. However, the current impact is difficult to evaluate because of other simultaneous factors such as incidental take in commercial fisheries. Nonetheless, there is documentation of recent egg use causing declines (Cornelius et al. 2007). From México to Colombia, olive ridley eggs have been and still are used for personal and commercial use (Lagueux, 1989; Arauz, 2000; Campbell, 2007; Cornelius et al., 2007). Laws regulating turtle egg use vary among 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. On unprotected solitary nesting beaches (most are unprotected), egg extraction often approaches 100%. Human use of turtle eggs for consumption and domestic animal consumption historically was widespread in the Indian Ocean and continues today largely wherever ridleys nest (Cornelius et al., 2007). Egg use has been reported in India, Bangladesh, Myanmar, Sri Lanka, Andaman Islands, Pakistan and Malaysia and is believed to have caused the decline of olive ridleys in these countries (Cornelius et al, 2007). Even at monitored beaches a proportion of the eggs are still lost to poaching.

Directed take of adults. In the East Pacific, although olive ridley turtle fisheries are now closed, illegal take of adult turtles still occurs widely with an unknown level of impact. There is evidence that thousands of olive ridleys are still taken each year along the Pacific coast of México (Frazier et al., 2007). In the West Atlantic, the direct take of adults has diminished over time to negligible levels (Cornelius et al., 2007). In the Indian Ocean, the use of adult olive ridleys and their eggs for personal use has been and continues to be widespread (Frazier, 1982; Frazier et al, 2007), and market-driven harvesting of eggs and females from nesting beaches are considered the greatest threat (Cornelius et al., 2007). Personal, subsistence use of adult olive ridley turtles is widespread worldwide (Cornelius et al., 2007; Frazier et al.,

2007). Olive ridleys and/or their eggs are used along the entire coast of West Africa (including Macaronesia) and sold in local and regional markets (Fretey, 2001).

Bycatch in fisheries.

The incidental capture of olive ridleys occurs worldwide in trawl fisheries, longline fisheries, purse seines, gill net 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 by-catch statistics are unavailable from fisheries, cause and effect has been used to implicate a fishery in the decline of olive ridleys. The incidental capture of olive ridleys in the shrimp trawl fishery in the western Atlantic, is believed to be the main cause of the significant population decline observed there since the 1970s and currently the number of olive ridleys by caught in trawl fisheries off the coasts of Surinam and French Guiana is believed to be approximately a couple of thousand turtles annually (Godfrey and Chevalier, 2004; Frazier et al., 2007). Gillnets and other fishing methods in this region also capture olive ridleys incidentally but to a lesser extent than shrimp trawl fishery (Frazier et al., 2007). Bycatch in trawl fisheries off Sergipe State in Brazil is considered the most pressing threat to that population (Thomé et al., 2003). In the eastern Atlantic, the incidental capture of olive ridleys by commercial fisheries is thought to be a significant threat but very little systematic data is available (Frazier et al., 2007). Incidental mortality of olive ridleys is worst along the coast of Orissa, India with arribada olive ridleys gathering to nest where fishing effort is high. Every year since the early 80s, thousands or tens of thousands of olive ridleys have stranded dead on the Orissa beaches, presumably as a result of incidental capture in shrimp trawls (Pandav, 2000). A gill net fishery also operates in the region and contributes to the ridley mortality along this coastline. Incidental capture in fisheries is also believed to be a serious threat in the eastern Pacific (Frazier et al, 2007) where olive ridleys aggregate in large numbers off shore from nesting beaches (Kalb et al., 1995; Kalb, 1999), but the information available is incomplete (Pritchard and Plotkin, 1995; NMFS/USFWS, 1998). Incidental capture of olive ridleys in this region has been documented in shrimp trawl fisheries, longline fisheries, purse seine fishery and gill net fisheries (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 annual capture for all species of marine turtle exceeding 60,000 turtles, most of which are olive ridleys (Arauz, 1996). Recent growth in the longline fisheries of this region are also a serious and growing threat to olive ridleys and have the potential to capture hundreds of thousands of ridleys annually (Frazier et al, 2007). Bycatch of olive ridleys is high in Indonesian tuna long-lines and shrimp trawls although mortality appears to be low (WWF Indonesia, unpublished data)

Habitat impacts.

Degradation, transformation and destruction of natural conditions at nesting beaches from coastal developments continue to threaten the long-term survival of many olive ridley rookeries. Transformation of nesting habitat comes from the construction of new aquaculture ponds, fishing harbors and tourist facilities, as well as growth of existing coastal villages which are increasing in many parts of the world within the range of the olive ridley, particularly along the east coast of India (Pandav and Choudhury, 1999) and in some zones in coastal México to Central America (Cornelius et al., 2007). These impacts contribute stress directly through the loss of nesting habitat or indirectly through changes in the thermal profiles of the beach, increased light pollution (Witherington 1992) and sewage effluents.

Global warming has the potential to impact the habitats and ecosystems of olive ridley populations worldwide (Hays et al., 2003; Weishampal, 2004) but the specific impacts are

purely speculative at this time. Most accounts have focused on the impact of global warming on incubation temperatures of eggs, which influence the sex ratio of the embryos (Hays et al., 2003).

Diseases and predation. Extremely little is known about diseases and their effects on olive ridley abundance. The only disease identified in the literature for olive ridleys is fibropapilloma, a herpes-virus found in sea turtles nearly worldwide (Herbst, 1994). The incidence of fibropapilloma is not believed to be high in olive ridleys but has been observed in olive ridleys nesting in Costa Rica (Herbst, 1994; Aguirre et al., 1999) and in México (Vasconcelos et al., 2000). At some individual rookeries, the predation by wild pigs and/or feral dogs can be substantial (e.g. in the Andaman Islands; Andrews et al., 2001). Infestation of developing eggs by fly and beetle larvae can cause significant mortality of embryos. In an extremely worrying case, the beetle larvae (*Omorgus suberosus fabricius*) has become a plague in the world's largest arribada rookery in Escobilla, México where it is provoking steep drops in the hatching efficiency of the clutches laid, from a typical 30% for this colony (Márquez, 1990) to less than 5% in some areas (López-Reyes and Harfush, 2000). When combined with the relatively low hatching rates of high-density arribada beaches and the destruction of eggs laid by previous nesters, this problem could provoke the rookery's decline!

Conservation Measures Taken and Required:

Most of the conservation actions on behalf of the olive ridley at national and international levels have been based on the species' listing under the *endangered* category in the IUCN Red List. As an Appendix I species under CITES (Convention on International Trade in Endangered Species) the international trade of skins from the species which fueled the large scale commercial exploitation of the olive ridley in from the 60s into the 80s was effectively halted. Other relevant international instruments that list the olive ridley as endangered and hence obligate its conservation by member states include: the Convention on Migratory Species (CMS) and the Inter-American Convention for the Protection and Conservation of Sea Turtles (IAC). CMS-promoted Memoranda of Understanding for the conservation and management of marine turtles and their habitats have been signed by the olive ridley's range states in the Indian Ocean and South-East Asia (known as IOSEA) as well as in other regions such as the Atlantic Coast of Africa under the Memorandum of Understanding concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa where 21 out of 26 range states participate.

On the basis of the species' classification in the IUCN Red List or in national endangered species lists, local legislatures of range states confer protection to the olive ridley. Although this sanctions law-enforcement, the implementation remains patchy at the global scale because of paucity in enforcement capabilities. Successful conservation has usually relied on well coordinated national programs in combination with local and non-governmental organizations incorporating public outreach. Statutory use and enforcement of the Turtle Excluder Devices in the shrimp trawlers has also proven critical in some areas with high levels of interaction with this fishery.

Despite the legislative efforts to protect the olive ridley, human impacts continue to be significant. In some areas (such as West Africa and South East Asia), extensive monitoring needs to be implemented to identify regions and stressors requiring priority actions. Bycatch and illegal take particularly from the coastal, artisanal fisheries need to be evaluated through adequate on-board observer programs and properly addressed. The beetle infestation of the Escobilla rookery must be adequately evaluated and acceptable measures of biological control of the insect need to be implemented. The impact from the increasing development of

much of the range state's coastline has to be evaluated and suitable mitigation measures implemented.

Red Listing

Red List Assessment: **Vulnerable**

Red List Criteria: A2bd

Assessment procedure. In accordance with the IUCN guidelines for Red List Assessments, the focus of this evaluation has been the number of mature individuals (IUCN, 2001). For olive ridleys, as with other sea turtle species, as it is not possible to survey mature individuals we used an index of abundance in the form of the number of *annual nesting females*. Although not all females breed every year (see Reproduction) and males are not evaluated, this index is considered to be the most reliable estimator for long-term population trends of marine turtles since the proportion of the total number of females that nest in any given year and the sex ratio is believed to be fairly constant across time within and between subpopulations (Meylan, 1982; Limpus, 1996).

Direct counts of the number of nesting females (NF) are not always available so we also relied on alternative information that can be converted to NF: number of nests per season, nests/km/yr or number of protected nests per season; annual estimates of hatchling or egg production, or census estimates of nesting females from arribada rookeries. When these proxies were used, the counted units were converted to NF based on the following constants: 105 eggs/nest and 2.5 nests/season/female. Some conservation programs reported annual number of protected nests which did not include the quantity of poached or otherwise destroyed or predated nests. For these cases we extrapolated to the total number of nests based on local estimates of conservation efficiency. All conversions to NF were made under the assumptions that (a) the mean number of eggs/nest and the number of nests/female/season do not differ significantly over the timescales we have used; and (b) the effort and the coverage of the biological parameters we used are reasonably constant through the time frame evaluated. In cases where the different surveys involved different levels of coverage, explicit corrections were made and specified in the result tables.

In spite of the olive ridley being the most abundant sea turtle, available quantitative information is extremely scarce and unevenly distributed across regions. We thus relied on a subset of the world's rookeries which, we assume, exhibit population trends that are representative of the population as a whole or, at least, for each of the regions. We selected 28 Index Sites (figure 1, table 1) for which reasonably long time series of quantitative data are available. They include all the largest known populations, as well as an assortment of smaller rookeries from almost all of the regions where the species is found. All sites are assumed to be demographically independent. However, although genetic data indicate a high degree of inter-rookery migration between some rookeries (e.g. Brazil-Suriname- Bowen et al., 1998; between México to Central America in the Eastern Pacific- Briseño-Dueñas, 1998) the results reflect events within an evolutionary timescale (many generations). Within the time span relevant to the assessment (2-3 generations), available evidence suggests significant demographic independence between the pairs of rookeries in question (e.g. mark-recapture in Nancite and Ostional indicate the vast majority are different turtles- reviewed by Bernardo and Plotkin, 2007; absence of exchange by tagged turtles and non-overlapping nesting seasons Brazil vs Suriname/French Guiana- de Silva et al., in press; and the lack of re-colonization of depleted arribadas in Jalisco and Guerrero in México by the very large Escobilla rookery, 500

to 1,000 kms distant in over four decades)

The population abundance estimates were based on raw data or on extrapolations from regressions performed on available data. As the relatively data-rich trends consistently indicated exponential trajectories we relied on these as the method of choice for extrapolations. Some data sets contained a number of trajectories across the time frames employed and in these more than one regression was employed or a combination of regressions and raw data. We have constrained our back extrapolations to time spans close to known historical events that are believed to have defined major abundance changes in order to avoid gross exaggerations with no supporting evidence. This has, however, meant that in some cases the extrapolations have only been performed back two generations (40 years) and could be considered underestimates of decline.

Uncertainties in the assessment process. Calculations based on very different datasets obtained by different workgroups, using multiple survey methods and spanning many decades are fraught with uncertainties. A number of these could be biasing our assessment. (1) Combining abundance information for individual rookeries obtained with a number of different methodologies could provide a source of error. However, we believe that the magnitude of these errors is of minor significance to the final declines estimated. (2) Because of the assessment's requirement for quantitative information, a very small proportion of the total known rookeries were included, and some regions are not well represented. The bias this introduces is further augmented since it is most likely that rookeries having long time series of data are also the most monitored and hence the better protected. In this case the estimates of decline will be underestimated relative to the true trends for the more numerous and less well monitored/protected rookeries. This is likely the case for regions with little (Indian Ocean and Western Pacific Ocean) or no (entire Eastern Atlantic) representation in our assessment that contribute very little to our global decline values yet where reports with qualitative evidence indicate extensive population declines (reviewed by Frazier et al., 2007; Cornelius et al., 2007). Nonetheless, although ideally a global assessment should incorporate full data from all regions to derive robust evaluations, the available information on the geographic distribution of abundances suggests that the largest rookeries are concentrated in regions with good representation and thus their inclusion will not significantly affect the global results. This is reflected at the regional level and in results for non-arribada rookeries where our sample bias will probably have caused an underestimation of true decline levels. (3) The extent of extrapolations into time past is an aspect that will undoubtedly contribute to the uncertainty of the results, particularly with scarce information. We have avoided extrapolating far beyond the oldest datasets to avoid this type of errors and thus consider that our results are conservative.

Rationale for the Red List Assessment:

In spite of scarcity in historical data, information from diverse sources has made it possible to evaluate a global decline for this widely distributed species over time periods ranging from decades to 2-3 generations. Striking regional differences are observed in the estimations which undoubtedly indicate far lower survival probabilities in some of the regions than what the global results would suggest.

There was also a stark and recurring contrast between the decline estimates for subpopulations grouped according to breeding strategy- arribada or non-arribada (solitary). The global decline rate estimated from non-arribada subpopulations (-63 to -83%) reflects a widespread low conservation status for these types of subpopulations that suggests they haven't recovered to historical (pre 1960) levels even in regions with long-term protection programs (e.g., over four decades in México) in spite generalized increments over the last decade

(Márquez et al., 1998). This needs to be highlighted because rookeries with non-arribada behavior are many times more numerous than those that nest as arribadas (e.g. in México about 98% of olive ridley subpopulations are non-arribada). Yet, as their abundances are up to 3-4 orders of magnitude smaller than arribada rookeries, they have an insignificant influence on the global decline estimates. In fact, the global net decline for the olive ridley is driven principally by population trends in just two arribada populations, Escobilla (México) and Ostional (Costa Rica), both in the Eastern Pacific.

The global decline value estimated on the basis of estimated population reductions of the annual number of olive ridley nesting females at subpopulations in the Index Sites used ranged between **28 and 34%** (Table 3). As most of the back extrapolations were limited to two generations it is likely that these values are conservative and hence we consider appropriate to accept the higher value.

When deciding whether to apply Red List decline criteria *A1* (the causes of reduction are clearly reversible AND understood AND ceased) or *A2* (the causes of reduction may not have ceased OR may not be understood OR may not be reversible) to obtain decline thresholds for the listing process three characteristics of the species' decline need to be analyzed (SPWG, 2006): (1) is the reduction reversible?, (2) are the causes of the reduction identified and understood?, and (3) have the threatening factors ceased? Since the decline estimated is driven by result from arribada rookeries, the questions need to be addressed against what is known for these types of populations. While it would appear that the elimination of large scale commercial exploitation of the olive ridley for leather and local consumption has allowed for the stabilization of a significant portion of rookeries, particularly in the Eastern Pacific and in particular facilitated the growth of arribada rookeries such as Escobilla and Ostional, the population growth of Mismaloya, Talcoyunque and Chacahua in the same region and under similar conservation circumstances remain at reduced abundances well below an arribada category. This could indicate that under some circumstances, the reduction of arribada rookeries below a certain level can make it impossible or unlikely for it to recover an arribada behavior. The major cause for the reduction in the species is thought to have been the massive commercial overexploitations, particularly in the Eastern Pacific. Furthermore, we do not yet fully understand nor are able to manage other stressors, some intrinsic or at least due to interactions between the overcrowding of growing populations that equally provoke dramatic declines in arribadas such as that of Nancite in spite of decades of protection (Cornelius et al., 2007). Though commercial exploitation of olive ridleys for international markets has effectively been eliminated, at local levels significant factors continue to impact individual rookeries such as excessive egg exploitation (e.g. Isla Caña, Panama) or bycatch (such as in Orissa, India). These examples indicate that olive ridleys, under current circumstances, do not meet all of the conditions for *A1* and hence should be evaluated with criteria *A2*.

Under *A2*, the decline estimations obtained for the olive ridley turtle at a global scale correspond to the *vulnerable* IUCN Red List threshold (a decline of >30% but < 50%).

Current Population Trend: Declining

Date of Assessment: 10 April, 2007

Assessor(s): Alberto Abreu-Grobois, Pamela Plotkin

Reviewed by: Milani Chaloupka (MTSG Red List Assessment Steering Committee Chair), Rod Mast, Nicholas Pilcher (MTSG co-Chairs)

Appendices & data tables

Table 1. List of olive ridley turtle index sites used for the status assessment

No.	Index nesting site	Justification and references
EASTERN PACIFIC OCEAN		
<i>ARRIBADA ROOKERIES (current or historical)</i>		
3	México- El Playon de Mismaloya, Jalisco	One of the five former arribada rookeries in México that collapsed in the 1970s due to overexploitation (Marquez et al., 1976) with some historical quantitative information.
6	México -Ixtapilla, Michoacán	New olive ridley rookery that now nests in arribadas. Not known prior to the early 1990s
7	México- Piedra de Tlalcoyunque, Guerrero	One of the five former arribada rookeries in México that collapsed in the 1970s due to overexploitation (Marquez et al., 1976) with some historical quantitative information.
8	México – Chacahua, Oaxaca	One of the five former arribada rookeries in México that collapsed in the 1970s due to overexploitation (Marquez et al., 1976) with some historical quantitative information.
9	México- Escobilla, Oaxaca	One of the two arribada rookeries remaining in México. Showing significant increases such that now > 1,000,000 nests laid/yr (Centro Mexicano de la Tortuga, 2005; Alvabera, 2006)
12	Nicaragua – La Flor	One of the two remaining arribada rookeries in Nicaragua, with some quantitative information
13	Costa Rica- Nancite	One of two historical arribada rookeries in CR, showing signs of sharp declines (ref. Valverde et al., 1998)
14	Costa Rica- Ostional	Currently most important arribada rookery in CR, with continuous extraction of eggs (Chávez et al., in press)
15	Panama – Isla Cañas	Most southerly arribada rookery in E Pacific.
<i>NON-ARRIBADA ROOKERIES</i>		
1	México- El Verde, Sinaloa	Northern limit of nesting range for LO in E Pacific. Excellent monitoring records since 1974 (Rios et al. 2005; Briseño-Dueñas and Abreu-Grobois, 1994)
2	México – Platanitos, Nayarit	Selection of non-arribada rookeries in México spanning the entire length of the country to increase the geographical variety of trends from a region with extensive monitoring of olive ridley subpopulations. Included Ixtapilla, a subpopulation which appears to have originated recently.
4	México – Cuyutlán, Colima	
5	México – Maurata+Colola, Michoacan	
10	México – Pto Arista, Chiapas	
11	Guatemala	Added context for Central American non-arribada rookeries (Muccio, 1998)
CENTRAL & WESTERN PACIFIC OCEAN		
	Australia – Arnhem Land	Most significant collection of rookeries in Central Pacific (Limpus, unpublished doc.)
16	Malaysia - Terengganu	Long term records available.
17	Thailand- Thaimaung	Additional insight from an area with traditional use of eggs (Limpus, 1995)
18	Thailand- Ko Pharathong	
20	Thailand- Maikaw Beach	

21	Indonesia-Alas Purwo NP	Rare example of recently increasing non-arribada rookery.
EASTERN INDIAN OCEAN		
<i>ARRIBADA ROOKERIES</i>		
22	India – Gahirmatha + Rushikulya + Devi River mouth	One of the largest sets of arribada rookeries in the world with long term datasets (Shanker et al., 2003)
<i>NON-ARRIBADA ROOKERIES</i>		
23	Myanmar	Site of ancient commercial egg harvest (Maxwell, 1911)
24	Bangladesh- St Martin’s Island	Additional regional coverage; extensive history of trade in turtle products and egg collection (Cornelius et al., 2007)
25	India – Cuthbert Bay, Andaman & Nicobar Islands	Area of traditional use of turtle meat and eggs, with extensive predation by wild and feral animals (Cornelius et al., 2007). Few sites surveyed in the past.
26	India – Chennai (Madras)	Representative of area in the region with widespread extraction of nests by humans and animals (Shanker, 2003). 15 years of data available (Shanker 2003, Kachhapa)
WESTERN INDIAN OCEAN		
28	Pakistan - Hawkes Bay and Sandspit, Karachi	Only representative from WIO. History of use in Baluchistan, Pakistan of direct fishery for olive ridleys (Cornelius et al., 2007)
EASTERN ATLANTIC OCEAN		
29	None available with long term quantitative information	
WESTERN ATLANTIC OCEAN		
<i>ARRIBADA ROOKERIES</i>		
29	Suriname- Eilanti	Historically an arribada rookery and primary nesting ground for the Western Atlantic LO subpopulation (Reichart, 1993)
<i>NON-ARRIBADA ROOKERIES</i>		
30	French Guiana-	Very little historical data but new monitoring programs by WWF-France (L. Kelle, pers. com.) and local groups are evidencing increasing population size.
31	Brazil - State of Sergipe: Abaís, Pirambu, Ponta dos Mangues and northern portion of Bahia State: Sítio do Conde, Costa do Sauípe, Praia do Forte and Arembepe	Southern limit for species, good monitoring from 1982 onwards demonstrating a significant and growing rookery (da Silva et al., in press)

Table 2. Summary of available estimates of *Past* and *Present* population abundance (raw estimates) at rookeries of *Lepidochelys olivacea* selected as Index Sites for this assessment. Codes for data types: FH, number of females harvested; NF, nesting females; AE, census estimate of nesting females from arribada beaches; NN, number of nests; PN, protected nests (would not include poached, predated and otherwise lost); NK, nests/km; EP, egg production; and, EH, egg harvest. All values are annual means unless otherwise stated.

Index #	Subpopulation	Data type	Past estimate 1		Past estimate 2		Present estimate		References (Past)	References (Present) and comments
			years	mean	years	mean	years	mean		
EASTERN PACIFIC										
<i>Arribada rookeries (current and former)</i>										
3	México (Playon de Mis-maloya, Jalisco)	AE / PN	1969 -1979	35,000 - 100,000 females			2001-2006	2,328 protected nests	Montoya, 1969; Márquez, et al 1976; Casas Andreu, 1978; Villa Guzmán, 1980	Antonio Trejo, in litt; Dirección General de Vida Silvestre. SEMARNAT México; (max. of last 5 years). Due to impossibility of complete protection assume only 60% of total nests protected
6	México (Ixtapilla, Michoacán)	NF / PN	Prior to 1994	0			1999-2005	2,900 - 10,000 nests	No known olive ridley rookery at this site prior to early 1990s	Dirección General de Vida Silvestre. SEMARNAT México (max. 5 years). Due to impossibility of complete protection assume only 35% of total nests reported
7	México (Piedra de Tlalcoyunque, Guerrero)	AE / PN	1974	20,000 – 50,000 females			1997	608 protected nests	Márquez, et al 1976	Dirección General de Vida Silvestre. SEMARNAT México; Márquez, et al 1998; Peñaflores et al., 2001. Due to impossibility of complete protection

			Past estimate 1		Past estimate 2		Present estimate			
										assume only 35% of total nests are protected
8	México (Chacahua, Oaxaca)	AE / PN	1974	20,000 – 50,000 females			2001-2005	2,042 nests	Márquez, et al 1976	JC Padilla (in litt.); Dirección General de Vida Silvestre. SEMARNAT México. 100% of nests are reported (max. of last 5 years)
9	México (Escobilla, Oaxaca)	AE / AE	1973-1975	180,000 - 401,000 females			2001-2005	1,013,034 females	Márquez, et al 1976 (arribadas in Aug-Oct only)	Dirección General de Vida Silvestre. SEMARNAT México; Alvabera, 2006 (max. of last 5 years)
12	Nicaragua (La Flor)	NN	1990s	28,000 - 48,000 nests			2002-2006	65,000-190,000 nests	Hope, 2002; Honarvar and van der Berghe (in press)	Urteaga and Motha (eds. 2007)
13	Costa Rica (Nancite)	AE / AE	1970s	142,000 - 335,000 females			2005	20,800 females	Valverde et al., 1998	Solís et al. (2007)
14	Costa Rica (Ostional)	AE / AE	1971	10,000 females			2006	336,000 females	Hughes and Richard, 1974; Chaves et al. (in press)	Solís et al. (2007)
15	Panama (Isla Cañas)	AE / AE	1990s	15,000 – 60,000 females			2006	8,768 females	Evans and Vargas, 1998	L. Vargas, Autoridad Nacional del Ambiente (in litt.) 2007
<i>Non-arribada rookeries</i>										
1	México (El Verde, Sinaloa)	NF / PN	1974	10,000-20,000 females			2000-2005	1,160 protected nests	Márquez et al., 1976	D. Ríos, Annual Reports 2000-2005; Dirección General de Vida Silvestre. SEMARNAT México

			Past estimate 1		Past estimate 2		Present estimate			
										(max. 5 years). Due to impossibility of complete protection only 90% of total nests are reported.
2	México (Platanitos, Nayarit)	NF / PN	1974	5,000-10,000 females			2000-2005	1,301 nests	Márquez et al., 1976	Dirección General de Vida Silvestre. SEMARNAT México. Due to impossibility of complete protection assume only 35% of total nests reported (max. 5 years);
4	México (Cuyutlán, Colima)	NF / PN	1974	5,000-10,000 females			1999-2003	1,257 nests	Márquez et al., 1976	Dirección General de Vida Silvestre. SEMARNAT México (max. 5 years);
5	México (Maruata-Colola, Michoacán)	NF / PN	1974	5,000-10,000 females			1999-2003	4,198 nests (max in 1993)	Márquez et al., 1976	Sum of two beaches; Dirección General de Vida Silvestre. SEMARNAT México (max. 5 years); Due to impossibility of complete protection assume only 35% of total nests reported
10	México (Pto Arista, Chiapas)	NF / NN	1974	1,000-5,000 females			1999-2004	707 nests	Márquez et al., 1976	Dirección General de Vida Silvestre. SEMARNAT México (max. 5 years); Due to impossibility of complete protection assume only 35% of total nests reported
11	Guatemala	NK	1981	1.87 nests/km/d			1997	1.24 nests/km/d	Muccio, 1998 from data in Higginson, 1982	Muccio, 1998. For 16 km Barra de Chapeton to Monterrico

			Past estimate 1	Past estimate 2	Present estimate					
CENTRAL & WESTERN PACIFIC OCEAN										
16	Malaysia (Terengganu)	EH / nests	1977	240,000			1998-1999	10	Malaysian Fisheries Dept data cited in Limpus , 2001	Liew, 2002
17	Thailand (Thaimaung)	NF	1975	300 females			1996-2000	10 nests	Chantrapornsyl, 1992	Chantrapornsyl, in litt. To A. Abreu
18	Thailand (Pharathong Island)	NF	1975	300 females			1996-2000	10 nests	Chantrapornsyl, 1992	Aureggi, 2001 cited by Chantrapornsyl, in litt. to A. Abreu
20	Thailand (Maikaw Beach)	NF	1975	150 females			1996-2000	10 nests	Chantrapornsyl, 1992	Chantrapornsyl, in litt. To A. Abreu
21	Indonesia (Alas Purwo NP)	NN	1984	10 nests			1993-1998	230 nests	Dermawan, 2002.	Dermawan, 2002.
EASTERN INDIAN OCEAN										
<i>Arribada rookeries</i>										
22	India (Gahirmatha + Rushikulya + Devi River mouth)	NF	1975	150,000 females			1990s - present	150-200,000 females	Bustard, 1976	Shanker et al., 2003
<i>Non-arribada rookeries</i>										
23	Myanmar	EP / NN	1911	1.5 mill eggs			1999	700 nests	Maxwell (1911)	Thorbjarnarson et al. , 2000
24	Bangladesh (St. Martin's)	NF	1980s	35 nesting females/night			2001	7 nesting females/night	S. M. A. Rashid, unpublished data, cited in Cornelius et al., in press	Islam, 2002

			Past estimate 1		Past estimate 2		Present estimate			
25	India (Andaman & Nicobar Islands)	NF / NN	1978-1989	445 nesting females			2001	185 nests	M. Tiwari report from Bhaskar, 1993	K Shanker, unpubl report
26	India (Chennai / Madras)	NK/NK	1977	~ 10 nests/km			2002	11.2 nests/km	Shanker 2003	Shanker, 2003
WESTERN INDIAN OCEAN										
28	Hawkes Bay, Pakistan	NN	1982-1984	25-120 nests			1996-1997	2 nests	Asrar, 1999	Asrar, 1999
WESTERN ATLANTIC OCEAN										
<i>Arribada rookeries</i>										
29	Suriname (Eilanti beach)	FH / NN	1967	2,875 females			2005	138 nests/yr	Geijskes (1945, cited in Reichart and Fretey 1993)	STINASU; Foundation for Nature Conservation and WWF Guianas
<i>Non-arribada rookeries</i>										
30	French Guiana	NN	1987	452 nests			2005-2006	2,600 nests	Fretey (1989) cited in Godfrey & Chevalier, unpublished report	Benoit de Thoisy Kwata NGO pers. comm..
31	Brasil (Sergipe)	NN	1940s	250 nests			2005-6	2,971 nests	Godfrey & Chevalier, unpublished report	da Silva et al., in press; L. Soares, in litt.

Table 3. Summary of estimates of population change for the olive ridley turtle Index Sites as determined from raw data and extrapolations from it for past abundances using procedures indicated in individual entries of the table. Past and Present estimates are provided in Table 2. Subpopulation size units were based on any of the following estimates: mean annual number of nesting females or of nests at site, or observations per unit effort. Unless otherwise stated, conversions from Table 2 data to number of females was determined using a mean value of 2.5 nests / female and 105 eggs / nest for any given nesting season. Extrapolation functions were used only when there was a suspected change in the subpopulation size over a time interval falling outside of the period covered by available raw data. When using regressions to extrapolate past abundances due to absence of information over the time interval required, linear (L) and/or exponential (E) functions (unless otherwise indicated) were employed using as much of the available abundance data as possible.

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
EASTERN PACIFIC OCEAN							
<i>Arribada rookeries (current and former)</i>							
3	México (Playon de Mismaloya, Jalisco)	35,000-100,000 females/yr (1969-1970)	2,328 protected nests (2001-2006)	1945-1965: assume no significant change 1965-1980: significant decline due to intense commercial extractions in Mexican Pacific of 1.5 -2.5 million adults (1960-1980), or 150,000 – 250,00 per year (Peñaflores et al., 2001) 1980-1990: continued extraction at levels 20% lower (derived from Peñaflores et al., 2001 data) (Depleted but <u>stable</u>; <u>no arribadas currently</u>)	341,204 (E. using 1969-1979 data to 1965)	3,024	-99% (2 generations)

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
6	México (Ixtapilla, Michoacán)	256 nests (1994)	2,853 nests	Olive ridley not known at this site prior to 1994, used earliest datum available	293 (taking into account 35% survey coverage)	3,261 –11,429 (taking into account 35% survey coverage)	3,806% ($\frac{1}{2}$ generations)
7	México (Piedra de Tlalcoyunque, Guerrero)	20,000 – 50,000 females (1974)	608 protected nests/yr (1997)	1945-1965: assume no significant change 1965-1980: significant decline due to intense commercial extractions in Mexican Pacific of 1.5 -2.5 million adults (1960-1980), or 150,000 – 250,00 per year (Peñaflores et al., 2001) 1980-1990: continued extraction at levels 20% lower (derived from Peñaflores et al., 2001 data) 1990-2005: stable (Depleted but stable; currently no arribas)	247,558 (E, using 1974-1993 data to 1965)	1,266 (E, using >1990 data and assuming 35% survey coverage)	-99.5% (2 generations)
8	México (Chacahua, Oaxaca)	20,000 – 50,000 females (1974)	2,042 nests (2001-2005)	1945-1965: assume no significant change 1965-1980: significant decline due to intense commercial extractions in Mexican Pacific of 1.5 -2.5 million adults (1960-1980), or 150,000 – 250,00 per year (Peñaflores et al., 2001) 1980-1990: continued extraction at levels 20% lower (derived from Peñaflores et al., 2001 data) 1990-2005: declining (Depleted and declining; currently no arribas)	98,563 (E, using 1974-1992 data to 1963)	460 (E, for 2005)	-99.5% (2 generations)

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
				<u>badas</u>)			
9	México (Escobilla, Oaxaca)	180,000 - 401,000 nesting females (Aug-Oct 1973-1975)	1,013,034 nesting females (2001-2005)	1945-1965: assume no significant change 1965-1980: significant decline due to intense commercial extractions in Mexican Pacific of 1.5 -2.5 million adults (1960-1980), or 150,000 – 250,00 per year (Peñaflares et al., 2001) 1980-1990: continued extraction at levels 20% lower (derived from Peñaflares et al., 2001 data) 1990-2005: increasing (Increasing and has incremented number of arribadas per year)	254,208 (E, extrapolation for data 1973- 1978) correcting 73-75 datum of 3 months by dividing by 0.68, the proportion of females in those months relative to whole year; Alva-bera, 2006)	574,937 (correcting for nest frequency)	+126% (2 generations)
12	Nicaragua (La Flor)	28,000 - 48,000 nests (1993-1999)	65,000 – 190,000 nests for La Flor (2002-2006)	1945-1970s: assume no significant change 1970-1980s: probably populations had decreased due to commercial extractions (Cornelius, 1982) but no quantitative information available. 1990s-2006: national reports indicate increases in La Flor (Depleted? and increasing)	33,441 (avg 1993-1999)	105,000 (avg 2002-2006)	+214 to +300% in last 13 years
13	Costa Rica (Nan-cite)	142,000 - 345,000 females (1970s)	20,800 females (2005)	1945-1980s: assume no significant change 1980s-1990: decline (Valverde et al., 1998) 1990s-2005: continued decline (Depleted? and declining)	138,000 (max value 1971-1981) 187,178 (E, regression to 1965)	8,320 corrected for nest frequency	-94 to -96%

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
14	Costa Rica (Ostional)	10,000 nesting females (1971)	336,000 nesting females (2006)	1945-1980s: assume no significant change. 1980s-2005: increasing? Will need to continue with Gates and Valverde method to determine if increasing or stable (at least stable and has incremented number of arribadas per year; Chávez et al in press)	4,000 correcting for nest frequency (1971 datum)	134,400 (correcting for nest frequency)	+3,260%
15	Panama (Isla Cañas)	15,000-60,000 nesting females (1997)	8,768 nesting females (2006)	1945-1970s: assume no significant change 1970s-1980: No long term information, though heavy egg extraction during this period led Central American coastal residents and government officials to acknowledged by early 1980s a decline in olive ridleys due to widespread egg harvesting augmented by the incidental capture of turtles in shrimp trawls and by the commercial turtle fishery in Ecuador (Cornelius, 1982). 1980s: high levels of turtle exploitation and poaching in island, increases in the fishing effort in nearby port. anecdotic evidence that arribadas at Isla Cañas have decreased in number (J.A. Cordoba, in litt and in Cornelius et al., 2007)	6,000-24,000 nesting females (accounting for 2.5 nest frequency)	3,507 nesting females (accounting for 2.5 nest frequency)	-42 to -85% (in ½ generation)
<i>Non-arribada rookeries</i>							

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
1	México (El Verde, Sinaloa)	10,000-20,000 nesting females (1974)	1,160 nests	1945-1965: assume no significant change 1965-1980: significant decline due to intense commercial extractions in Mexican Pacific of 1.5 -2.5 million adults (1960-1980), or 150,000 – 250,00 per year (Peñaflores et al., 2001) 1980-1990: continued extraction at levels 20% lower (derived from Peñaflores et al., 2001 data) 1990-2005: stable (Depleted but <u>stable</u>)	10,000 to 20,000 (consider these values to reflect fairly well pre-1965 abundances)	306-516 (assuming 90% survey coverage and 2.5 nest frequency)	-95 to -97% (2 generations)
2	México (Platanitos, Nayarit)	5,000-10,000 nesting females (1974)	1,301 nests	1945-1965: assume no significant change 1965-1980: significant decline due to intense commercial extractions in Mexican Pacific of 1.5 -2.5 million adults (1960-1980), or 150,000 – 250,00 per year (Peñaflores et al., 2001) 1980-1990: continued extraction at levels 20% lower (derived from Peñaflores et al., 2001 data) 1990-2005: increasing (Depleted but <u>increasing</u>)	5,000 to 10,000 (consider these values to reflect fairly well pre-1965 abundances)	439-1,301 (assuming 35% survey coverage and 2.5 nest frequency)	-74 to -87% (2 generations)

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
4	México (Cuyutlán, Colima)	5,000-10,000 nesting females (1974)	4,198 max nests (1999-2003)	1945-1965: assume no significant change 1965-1980: significant decline due to intense commercial extractions in Mexican Pacific of 1.5 -2.5 million adults (1960-1980), or 150,000 – 250,00 per year (Peñaflores et al., 2001) 1980-1990: continued extraction at levels 20% lower (derived from Peñaflores et al., 2001 data) 1990-2005: increasing (Depleted but <u>increasing</u>)	5,000 to 10,000 (consider these values to reflect fairly well pre-1965 abundances)	483-1,437 (assuming 35% survey coverage)	-71 to -86% (2 generations)
5	México (Maruata-Colola, Michoacán)	5,000-10,000 nesting females (1974)	4,198 nests (max in 1993)	1945-1965: assume no significant change 1965-1980: significant decline due to intense commercial extractions in Mexican Pacific of 1.5 -2.5 million adults (1960-1980), or 150,000 – 250,00 per year (Peñaflores et al., 2001) 1980-1990: continued extraction at levels 20% lower (derived from Peñaflores et al., 2001 data) 1990-2005: stable (Depleted but <u>stable</u>)	5,000 to 10,000 (consider these values to reflect fairly well pre-1965 abundances)	591 -2,806 (assuming 35% survey coverage)	-44 to -72% (2 generations)

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
10	México (Pto Arista, Chiapas)	1,000-5,000 females (1974)	707 nests	1945-1965: assume no significant change 1965-1980: significant decline due to intense commercial extractions in Mexican Pacific of 1.5 -2.5 million adults (1960-1980), or 150,000 – 250,00 per year (Peñaflores et al., 2001) 1980-1990: continued extraction at levels 20% lower (derived from Peñaflores et al., 2001 data) 1990-2005: stable (Depleted but <u>stable</u>)	1,000 to 5,000 (consider these values to reflect fairly well pre-1965 abundances)	85 - 808 (taking into account 35% survey coverage)	-19 to -84% (2 generations)
11	Guatemala	1.87 nests/km/d (1981)	1.24 nests/km/d (1997)	1945-1960s: assume no significant change 1960s-1980: heavy egg extraction; Central American coastal residents and government officials acknowledged by early 1980s a decline in olive ridleys due to widespread egg harvesting augmented by the incidental capture of turtles in shrimp trawls and by the commercial turtle fishery in Ecuador (Cornelius, 1982). 1990s-present: legal harvests probably unsustainable as it only protects <2% of all eggs (Muccio, 2000)	3,191 (E, extrapolated to 16km and all season to 1960)	1,004 (E, extrapolated to 16km and all season to 2005)	-69% (2 generations)

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
CENTRAL & WESTERN PACIFIC OCEAN							
16	Malaysia (Terengganu)	240,000 eggs harvested/yr (1977)	<10 nests	Common local practice of egg harvesting in large numbers at least since the 1950s. Practically extinct.	813 (L, extrapolated to 1950)	<4	-99.5%
17	Thailand (Thaumaung)	300 nesting females (1975)	10 nests (1996-2000)	Probably long-term egg harvest (Limpus, 1995). Assume started in the 1950s so back extrapolation only to 1950	238 (L; correcting for nest frequency)	4	-98%
18	Thailand (Pharathong Island)	300 nesting females (1975)	10 nests (1996-2000)	Probably long-term egg harvest (Limpus, 1995). Assume started in the 1950s so back extrapolation only to 1950	238 (L; correcting for nest frequency)	4	-98%
20	Thailand (Maikaw Beach)	150 nesting females (1975)	10 nests (1996-2000)	Probably long-term egg harvest (Limpus, 1995). Assume started in the 1950s so back extrapolation only to 1950	118 (L)	4	-97%
21	Indonesia (Alas Purwo NP)	10 nests (1984)	230 nests max (1993-1998)	Unknown if depleted from earlier times	4	92	+2,200%
EASTERN INDIAN OCEAN							
<i>Arribada rookeries</i>							
22	India (Gahirmatha + Rushikulya + Devi River mouth)	150,000 nesting females	150,000 – 200,000 nesting	Commercial harvest of sea turtle eggs occurred since colonial days at Gahirmatha and continued until 1974 when it became illegal	150,000 – 200,000 (did not need correcting for nest fre-	150,000 – 200,000 (did not need	0%

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
		(1975)	females (1990s-present)	(Kar, 2001). Up to 1,500,000 eggs a year were harvested (e.g. in 1973, FAO, 1974), although the illegal take was probably much more (Kar, 1988, 2001). In 1974, 800,000 eggs were collected (FAO, 1974), after which all licensed harvesting was halted (Kar, 2001). Nevertheless, and in spite of death of about 15,000 breeding adults per year from the mid 1990s to the present (Shanker et al., 2003) no consistent evidence of decline in the set of 3 rookeries (Shanker et al., 2003). Therefore assume stable over last 3 generation period.	quency)	correcting for nest frequency)	
<i>Non-arribada rookeries</i>							
23	Myanmar	1.5 mill eggs harvested (1911)	700 nests (1999)	Continuous egg harvest since early 1900s. Assume steady linear decline from 1911 datum to present	3,429 (L)	280	-92%
24	Bangladesh (St. Martin's)	35 nesting females/night (1980s)	7 nesting females/night (2001)	With no protection, by the 1980s anecdotal accounts underline declines due to severe exploitation of eggs and illegal killing of adult turtles by fishing and other activities (Islam 1998). Assume same rate of linear declines since the 1960s.	62 (L)	7	-89%
25	India (Andaman & Nicobar Islands)	445 nesting females (1978-	185 nests (2001)	Unknown historical abundances or trends. Use available information that nonetheless indicates significant decline in less than three generations.	178 (correcting for nest frequency)	74	-58% in less than 1 generation

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
		1989)					
26	India (Chennai / Madras)	~ 10 nests/km	11.2 nests/km	Substantial fluctuations in abundance, from 3.7 – 14.3 nests/km between 1988 - 2002			0
WESTERN INDIAN OCEAN							
28	Pakistan - Hawkes Bay and Sandspit, Karachi	25-120 nests (1982-1984)	2 nests	Unknown historical abundances or trends. Use available information that nonetheless indicates significant decline in less than three generations.	10-48 in 1980s	< 1	-60 to -98% (1 generation)
WESTERN ATLANTIC OCEAN							
<i>Arribada rookeries</i>							
29	Suriname (Eilanti beach)	2,875 females (1967)	411 nests/yr (200s)	As per review by Godfrey & Chevalier (unpublished) take of nesting females goes back at least to the 1930s. Convert to 1945 data using assumption that min 15 and max 50% of all females were being harvested per year (1,500) and continuous linear decline (Godfrey and Chevalier, unpublished). Geijskes (1945, cited in Reichart and Fretey 1993) Also calculated using a back extrapolation from exponential regression of decline rate to derive 1945 and 1965 values (5,354 and	1,758 (L)	164	-91 to -99%
					5,647 (L)		

Index #	Subpopulation (Index Site)	Raw Data (from Table 2) Past	Raw Data (from Table 2) Present	Notes on Population Trajectories & (Comments on Current Status)	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size	% Change over time period
				1,125, respectively) so consider the above values acceptable. Note: as Eilanti beach does not exist anymore, the present monitored areas are not the same but we have assumed they represent remnants of the same population			
<i>Non-arribada rookeries</i>							
30	French Guiana	1,000 nests (1940s)	2,600 nests (avg 2005 & 2006)	No historical information, made best approximations based on oldest surveys available and most plausible past abundances (Godfrey and Chevalier, unpublished)	400 (L)	1,040 (correcting for nest frequency)	+160%
31	Brasil (Sergipe)	250 nests (1940s)	2,971 nests (2005-6)	No historical information, made best approximations based on oldest surveys available and most plausible past abundances (Godfrey and Chevalier, unpublished)	100 (L)	582 (correcting for nest frequency)	+1,396%
<i>global totals <u>only for non-arribada</u> sites (max / min)</i>					60,309	10,281	-67%
					31,271	5,319	-83%
<i>global totals <u>only for arribada</u> sites (max / min)</i>					1,252,028	855,403	-27%
					1,178,092	847,231	-32%
GLOBAL TOTALS (max / min)					1,312,337	865,684	-28%
					1,209,363	852,550	-34%

Table 5. Population changes for the olive ridley Index Sites grouped by region. Changes are determined as explained in previous sections to obtain compilations of regional subtotals.

Region	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size (2005)	% Change
EASTERN PACIFIC OCEAN			
<i>Arribada rookeries</i> (Playon de Mismaloya, Jalisco, México; Ixtapilla, Michoacán, México; Piedra de Tlalcoyunque, Guerrero, México; Chacahua, Oaxaca, México; Escobilla, Oaxaca, México; La Flor, Nicaragua; Nancite, Costa Rica; Ostional, Costa Rica; Isla Cañas, Panama)	1,170,381	779,343	-29% to -33%
	1,100,334	771,175	
<i>Non-arribada rookeries</i> (El Verde, Sinaloa, México; Platanitos, Nayarit, México; Cuyutlán, Colima, México; Maruata-Colola, Michoacán, México; Pto Arista, Chiapas, México; Barra de Chapeton - Monterrico, Guatemala)	58,191	7,871	-95% to -73%
	29,191	2,908	
<i>totals</i>	1,228,572	787,214	-30% to -37%
	1,129,525	774,083	
CENTRAL & WESTERN PACIFIC OCEAN (<i>only non-arribada rookeries</i> - Terengganu, Malaysia; Thaimaung, Thailand; Pharathong Island, Thailand; Maikaw Beach, Thailand; Alas Purwo NP, Indonesia)	1,412	108	-92%
EASTERN INDIAN OCEAN			
<i>Arribada rookeries</i> (Gahirmatha + Rushikulya + Devi River mouth, India)	76,000	76,000	0%

Region	Past Annual Nesting Female Subpopulation Size	Present Annual Nesting Female Subpopulation Size (2005)	% Change
<i>Non-arribada rookeries</i> (Myanmar; St. Martin's, Bangladesh; Andaman & Nicobar Islands, India)	178	74	-58%
<i>totals</i>	76,178	76,074	0%
WESTERN INDIAN OCEAN (<i>only non-arribada rookeries-Hawkes Bay, Pakistan</i>)	48	4	-98% to -60%
	10	1	
WESTERN ATLANTIC			
<i>Arribada rookeries</i> (Eilanti beach, Suriname)	5,647	55	-99% to -97%
	1,758	55	
<i>Non-arribada rookeries</i> (Cayenne and Kourou, French Guiana; Sergipe, Brasil)	480	2,228	364%
<i>totals</i>	6,127	2,284	-63% to 2%
	2,238	2,284	

Table 6. Country Distribution

	Country	presence	nesting	foraging	Reference
	Western Atlantic				
1	Cuba	P	F		Moncada et al., 2000
2	Puerto Rico	P	F		Caldwell and Erdman, 1969; Caldwell et al. 1969
3	Jamaica	P			Brongesma, 1972 in Fretey, 1999
4	Dominican Republic	P	N?		Bacon, 1981 in Fretey, 1999
5	Barbuda	P?			Fuller et al., 1992 in Fretey, 1999
6	Guadeloupe	P			Fretey and Lescure, 1999
7	Martinique	P			Bacon, 1981 Fretey and Lescure, 1999
8	Curaçao	P			Sybesma and Hoetjes, 1992 in Fretey, 1999
9	Panama	P?		F	Bacon, 1981 in Fretey, 1999
10	Trinidad and Tobago	P	N1	F	Carr 1957; Gyan, 1987 in Fretey, 1999
11	Venezuela	P	N1	F	Nowak, 1974 in Fretey, 1999
12	Guyana	P	N		Pritchard and Trebbau, 1984, Fretey, 1999; Godfrey and Chevalier, 2004
13	Suriname	P	N		Reichart and Fretey. 1993 Schulz, 1975 Caldwell et al. 1969, Pritchard & Trebbau 1984; Godfrey and Chevalier, 2004
14	French Guiana	P	N		Fretey, 1989 in Fretey, 1999; Godfrey and Chevalier, 2004
15	Brazil	P	N		Marcovaldi and Marcovaldi, 1987; da Silva et al., 2003
16	Uruguay	P			Frazier, 1984 in Fretey, 1999
	Eastern Atlantic				
17	Morocco	P?	N?	XF	Fretey, 2001
18	Mauritania	P?	XN	XF	Fretey, 2001
19	Cape Verde	P	XN	XF	Fretey, 2001
20	Senegal	P	XN	XF	Fretey, 2001
21	Gambia	P	XN	XF	Barnett et al., 2004
22	Guinea Bissau	P	N	XF	Barbosa et al., 1998
23	Guinea	P	N?	XF	Fretey, 2001
24	Sierra Leone	P	N	XF	Siaffa et al., 2003
25	Liberia	P	N	XF	Fretey, 2001
26	Cote D' Ivoire	P	N	XF	Gomez et al., 2003
27	Ghana	P	N	XF	Beyer 2002
28	Togo	P	XN	XF	Hoinsoude et al., 2003
29	Benin	P	N	XF	Doussou Bodjrenou et al., 2004
30	Sao Tome & Principe	P	N	XF	Fretey, 2001
31	Boiko, São Tome, Corisco, Mbanye, Hoco Islands				Fretey et al., 2004
32	Nigeria	P	N	XF	Fretey, 2001
33	Cameroon	P	N	XF	Fretey et al., 2004
34	Equatorial Guinea	P	N	XF	Fretey et al., 2004
35	Gabon	P	N	XF	Fretey et al., 2004
36	Congo	P	N	XF	Fretey et al., 2004
37	Angola	P	N	F	Fretey et al., 2004
38	Namibia	P	XN	XF	Fretey, 2001

39	South Africa	P	XN	XF	Fretey, 2001
	Red Sea				
40	Eritrea		N		Ross & Barwani, 1982;Frazier, 1975, 1980; Pilcher et al., 2006
	Western Indian Ocean				
41	India, West coast	P	N	?	Kar & Bhaskar, 1982; Shanker & Choudhury, 2006; Sunderraj et al, 2006
42	Pakistan	P	N	?	Kabraji & Firdous, 1984; Asrar, 1999; Qureshi, 2006
43	Iran	P	XN	?	Kami, 1997
44	Kenya	P	N	?	Zanre, pers. comm.; Church, 2004
45	Madagascar	P	XN	F	Frazier, 1975, 1980, 1982; Pritchard, 1979
46	Maldives	P	XN	?	Frazier, 1975, 1980, 1982; Pritchard, 1979
47	Mozambique	P	N	?	Frazier, 1975, 1980, 1982; Pritchard, 1979
48	Oman	P	N	?	Ross & Barwani, 1982; Frazier, 1975, 1980, 1982; Salm, R.V., Jensen, R.A.C. & Papastavrou, V.A. (1993); Baldwin. R.M. and A. A. Al- Kiyumi.(1999)
49	Somalia	P	?	?	Frazier, 1975, 1980, 1982
50	South Africa	P	N	?	Hughes, 1972
51	Sudan	P?	XN	X	Ross & Barwani, 1982; Frazier, 1975, 1980, 1982
52	Tanzania	P	N	?	Frazier, 1975, 1980, 1982; Frazier,1976
53	Yemen	P?	XN	?	Walczak, 1979
	Eastern Indian Ocean				
54	Andaman & Nicobar Islands	P	N		Andrews et al., 2006
55	Thailand	P	N	?	Aureggi et al., 2004
56	Bangladesh	P	N	?	Rashid and Zahirul. 1998; Islam, 2002a,b; Sarker, 2004; Rashid, 2006
57	India, East coast	P	N	F	Bhaskar, 1981; Kar & Bhaskar, 1982; Shanker & Choudhury, 2006; Tripathy et al., 2003; Bhupathy and Saravanan, 2006; Shanker, 2003
58	Myanmar	P	N	?	Thorbjornarsson et al, 2000; Shanker and Pilcher, 2003
59	Sri Lanka	P	N	F	Dattatri & Samarajiva 1983; Hewavisenthi, 1990; Amarasooriya and Jayathilaka, 2002; Kapurusinghe, 2006; De Silva, 2006
	Western Pacific				
60	Australia	P	N	?	Harris, A. 1994; Whiting, S. D. 1997; Limpus, C. J. Miller, J. D. and Fleay, A. 1981; Limpus, 1975
61	Brunei	P	N	?	Pilcher, 2001; Shanker and Pilcher, 2003
62	Cambodia	?	?	F	Kosal, Mao
63	Federated States of Micronesia	P		?	Falanruw, M. V. C. Mccoy, M. and Namlug. 1975
64	Indonesia	P	N	F	Suwelo, 1999; Putrawidjaja, 2000; Halim et al., 2001
65	Japan	P		?	Kamezaki, 1999
66	Malaysia	P	N	?	Tisen and Bali, 2002
67	Papua New Guinea	P	N	F	Spring, C. S. , 1979; Spring, C. S. and J. Gwyther, 1999; Phillip, 2002

68	Philippines	P	N	?	Gomez, E. D. 1980.
69	Taiwan	?			
70	Thailand	P	N	?	Chantrapornsyl, S. 1992; Charuchinda, M. and S. Chantrapornsyl. 1999
71	USA (Hawai'i)	P		?	Balazs and Hau, 1986
72	Vietnam	N		?	Shanker and Pilcher, 2003; Hamann et al., 2006
	Eastern Pacific				
73	EUA	P			Houck and Joseph, 1958; Hubbs, 1977
74	México	P	N	F	Márquez et al., 1976
75	Guatemala	P	N	F	Higginson, J. & Orantes R. 1987; Higginson, 1989; Márquez, 1990; Muccio, 1998; Juarez and Muccio, 1997
76	El Salvador	P	N	F	Cornelius, 1982; Hasbún & Vásquez. 1999
77	Honduras	P	N	F	Cornelius, 1982; Minarik, C.J. 1985; Lagueux, 1989; Lagueux, 1991
78	Nicaragua	P	N	F	Cornelius, 1982; Martinez, P. Gutierrez, C. and Arauz, R. 1998; Pritchard, 1979
79	Costa Rica	P	N	F	Hughes and Richard, 1974; Pritchard, 1979
80	Panama	P	N	F	Cornelius, 1982; Córdoba L. 2000; Pritchard, 1979
81	Colombia	P	N	F	Amorocho, et al., 1989; Martínez, 2000; Pritchard, 1979; Martinez and Paez, 2000
82	Ecuador	P	N	F	Hurtado, M. 1981; Green and Ortiz-Crespo, 1982; Fritts, T. H. 1981.; Alava et al. In press?
83	Peru	P	N	F	Brown and Brown, 1982
84	Chile	P			Brito, J. 1994.

1 Possibly now extinct or no recent evidence of continued nesting
x="no"

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