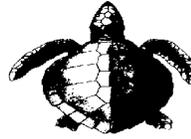


**STATUS REVIEWS OF SEA TURTLES LISTED UNDER
THE ENDANGERED SPECIES ACT OF 1973:**



Loggerhead Turtle, *Caretta caretta*
East Pacific Green Turtle, *Chelonia mydas*
Leatherback Turtle, *Dermochelys coriacea*
Hawksbill Turtle, *Eretmochelys imbricata*
Kemp's Ridley Turtle, *Lepidochelys kempii*
Olive Ridley Turtle, *Lepidochelys olivacea*



United States Department of Commerce
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
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U.S. Fish and Wildlife Service
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EXECUTIVE SUMMARY

Introduction.

Section 4 of the Endangered Species Act requires that National Marine Fisheries Service (NMFS) and the U.S. Fish and Wildlife Service (USFWS) review the status of listed species at least once every five years. The Services use these status reviews to determine whether a designation as threatened or endangered accurately reflects the current status of a listed species. If the status of the species has either improved or deteriorated, appropriate action will be taken to ensure that the species is listed accurately. The status reviews for the individual species of sea turtles were compiled by independent experts for NMFS and USFWS from 1990 through May 1992 and were updated by Pamela Plotkin, Editor, in 1995. The status review for the green turtle (*Chelonia mydas*) is incomplete and will be published separately. These status reviews represent the best biological information available to the expert authors.

Summary of Findings:

Loggerhead turtle (*Caretta caretta*). It is recommended that the loggerhead turtle remain listed as threatened throughout its range. Information indicates that its status has worsened in recent years. The numbers of nesting females have declined in Georgia and South Carolina, while Florida's nesting aggregations have remained intact. However, increased human presence in areas of heavy nesting is likely to impact dense nesting populations in Florida in the near future. Further study is needed on U.S. loggerhead population stock structure. Recent evidence suggests that the adult loggerheads nesting in Georgia may represent a population that is distinct from those turtles nesting in Florida. If there are two separate populations along the southeast Atlantic coast, then the northern one (i.e., those nesting in

Georgia, South Carolina, North Carolina, and northward) is in severe decline and should this continue, consideration must be given to listing them as endangered.

Leatherback turtle, (*Dermochelys coriacea*). It is recommended that the leatherback turtle remain listed as endangered throughout its range. While the leatherback turtle enjoys complete protection in the United States, the species is often inadequately protected elsewhere. Leatherback turtles foraging in the United States include juveniles which will eventually recruit into the breeding aggregations of the western Atlantic and eastern Pacific, where they will face considerable threats at the nesting grounds. In Mexico, the largest breeding colony of leatherback turtles in the western hemisphere has suffered enormous losses of both adults and eggs in recent decades. Similarly nesting leatherbacks were killed and eggs collected in Malaysia and the British Virgin Islands where commercial and subsistence take has extirpated once thriving populations. Threats to leatherbacks in the marine environment have not been satisfactorily quantified but appear serious. Large juveniles and breeding age adults are at increasing risk due to incidental capture and drowning in fishing gear and entanglement and ingestion of persistent marine debris.

Hawksbill turtle, (*Eretmochelys imbricata*). It is recommended that the hawksbill turtle remain listed as endangered. Depleted U.S. populations are not currently considered to be declining, but neither are there indications of recovery despite more than a decade of legal protection. In addition, habitat destruction and clear evidence of over-exploitation continue on a global scale. With the exception of the Kemp's ridley (*Lepidochelys kempii*), the hawksbill is considered by many to be the most endangered of all the marine turtles. There is little doubt that U.S. Caribbean and Pacific nesting colonies have been severely depleted during the twentieth century. The illegal domestic harvest of eggs and turtles continues in the United States, especially in Caribbean and Pacific island territories.

Kemp's ridley turtle, (*Lepidochelys kempii*). It is recommended that the Kemp's ridley turtle remain listed as endangered. The Kemp's ridley is the most endangered of all sea turtle species and shows little sign of recovery. A minor increase in nesting may be an artifact of greater nesting beach coverage or may indicate that the nesting population has increased. The lower average clutch size (95 eggs) that has been reported in recent years may indicate an influx of novice nesters. Efforts to restore the species must concentrate upon protecting subadult and adult animals and must take place over a long period of time because of the species' slow maturation. The key to this task is the use of turtle excluder devices throughout the U.S. and Mexican shrimp trawl fisheries in adult and developmental habitats, and continued protection of the species' nesting beach.

Olive Ridley Sea Turtle, (*Lepidochelys olivacea*). It is recommended that the olive ridley turtle in the eastern Pacific continue as threatened, the breeding population in Mexico continue as endangered, and the western Atlantic population be reclassified to endangered. The status of the olive ridley in the western Atlantic is unambiguous. Populations are very low and continue to decline, almost certainly as a result of long-standing incidental capture in shrimp trawls. This population is in worse condition than that of the endangered Kemp's ridley and should be listed as endangered. Available data are too few to assess the status of the species in the eastern Atlantic and northern Indian Oceans. However, probable widespread egg collection throughout West Africa and known problems with incidental capture and mortality in shrimp trawlers and gill nets near the arribada beaches of Orissa, India seem to indicate that the populations there are stressed and should continue to be listed as threatened. In the eastern Pacific, data indicate that some nesting aggregations are in decline, while others appear relatively stable. There currently are no controls at all on incidental capture by shrimp trawl and long-line fisheries throughout this vast region, which appears significant throughout much of the region.

Loggerhead Sea Turtle, *Caretta caretta*

Prepared by Nat B. Frazer, Ph.D.

Biological Background

Nesting adult female loggerheads in the southeastern U.S average about 92 cm in straight-line carapace length, with a mean mass of about 115 kg (NRC 1990, NMFS 1990). Thus, they tend to be larger than ridleys (*Lepidochelys olivacea* and *L. kempii*) and hawksbills (*Eretmochelys imbricata*) but smaller than green turtles (*Chelonia mydas*) and leatherbacks (*Dermochelys coriacea*). They have a carapace that is covered with horny scutes, as do the other members of the family Cheloniidae. Loggerheads typically have five vertebral scutes, with five costal scutes on either side of these, and eleven or twelve marginals lateral to the costal scutes. However, there is much individual variation in the number of scutes, and it is not unusual to find individuals displaying asymmetrical scute arrangements (e.g., eleven marginals on one side and twelve on the other). The nuchal scute is in contact with the first costal on either side, and the head is much larger in relation to the carapace than is the case with other members of the Cheloniidae. The tendency of many loggerheads to have an extensive community of epibionts covering the carapace (Caine 1986) often obscures their carapacial markings. Ventrally, there are almost invariably three pairs of pore-less inframarginal scutes on the "bridge" between the carapace and plastron. Adults and larger juveniles have a orange or reddish-brown carapace and a yellowish or light brown plastron. Hatchlings range in color from light brown to almost black, with a three-keeled carapace about 4.2 cm long and a somewhat lighter two-keeled plastron. The keels disappear as the turtles increase in size (age). More complete descriptions and a review of taxonomic and descriptive literature may be found in Dodd (1988).

Like other turtles in general (Wilbur and Morin 1988), and other sea turtles in particular (NRC 1990), loggerheads are relatively long-lived and late-maturing animals (Dodd 1988). Growth rates of loggerheads reared in captivity for fourteen years in semi-natural conditions indicate that maturity may be reached in about 16 years (Frazer and Schwartz 1984). However, additional captive studies indicate that growth rates are dependent upon temperature, food quantity and food quality (Stickney et al. 1976, Nuijta and Uchida 1982), casting some doubt as to the applicability of information obtained from studies of captive loggerheads in providing accurate estimates of growth rates and age at maturity in natural populations.

Based on studies of growth rates for wild loggerheads in Florida and Georgia waters, age at maturity is estimated to be from 10-30 years (Frazer and Ehrhart 1985, Zug et al. 1986). The maximum lifespan for females in the wild would then be about 62 years, assuming that it takes 30 years to mature, with a maximum of an additional 32 years of reproductive life-span thereafter (Frazer 1983). It is likely, however, that age at maturity and growth rates vary widely among, and even within, local populations of loggerheads (Dodd 1988), as has been demonstrated in long-term studies of freshwater turtles (Gibbons et al. 1981, Zweifel 1989, Frazer et al. 1991). Recent estimates on age at maturity of Australian loggerheads suggests that females mature between 34.3 and 37.4 years (Frazer et al. 1994).

Adult female loggerheads are iteroparous both within and among years, typically displaying high fecundity. Most females usually nest from three to five times in a season, although numerous individuals have been recorded laying six nests (Frazer and Richardson 1985a, LeBuff 1990) and seven nests have been recorded for one individual (Lenarz et al. 1981). Average clutch size differs according to location, but usually falls between 95 and 150 eggs per nest (Frazer and Richardson 1985b, Dodd 1988, LeBuff 1990). The estimated survival

rate from the time eggs are laid until the hatchlings reach adulthood is only about 0.1 - 0.2 % (i.e., only one or two per thousand: Frazer 1986a).

Although loggerheads display iteroparity among years, most adult females do not return to nest in successive years (Dodd 1988). The most frequently observed interseasonal nesting interval for loggerheads in the U.S. is 2 years, although the intervals are not well understood (Frazer 1989a) and individuals are known to increase or decrease their interseasonal nesting intervals (Frazer 1984).

It is not unusual to observe large fluctuations in numbers of nesting loggerheads from year to year in a given locality (NRC 1990), perhaps due to individuals shifting their interseasonal intervals. This is similar to the behavior of freshwater turtle populations in which the proportion of reproductively active females may fluctuate widely from year to year (Frazer et al. 1990). The causes for such fluctuations are not understood for any turtle species, but are presumed to be environmentally induced, perhaps involving the accumulation of resources (e.g. nutrient or energy reserves) necessary for reproduction (Wilbur and Morin 1988).

Loggerheads are circumglobal in their distribution, inhabiting coastal, estuarine and continental shelf waters in subtropical and tropical areas as well as in the temperate zones as far north as Newfoundland and as far south as Argentina (NRC 1990, NMFS 1990). Their major nesting areas are described as being "antitropical" (Pritchard 1979), indicating their tendency to nest in subtropical areas immediately north of the Tropic of Capricorn or south of the Tropic of Cancer. Some nesting occurs even further up into temperate areas such as Japan and the Mediterranean between 30° and 40° N latitude (Dodd 1988). However, there also is some scattered nesting of loggerheads in tropical areas such as the Caribbean, and what is perhaps the world's largest nesting aggregation (ca. 30,000 nesting females per year)

is on Masirah Island, Oman (Dodd 1988). Additional important nesting areas (> 200 females per year) are found in Australia, Brazil, Colombia, Greece, Jamaica, Mexico, Madagascar, Malagasy Republic, Mozambique, South Africa and the U.S. (Groombridge 1982, Mager 1985, Ehrhart 1989).

Loggerheads occur in Atlantic, Gulf of Mexico, and Caribbean waters under the jurisdiction of the United States (Ehrhart 1989). Nesting occurs from New Jersey (Brandner 1983) around the Gulf of Mexico to Texas (Shaver 1988) and to Puerto Rico in the Caribbean (Ehrhart 1989). The major U.S. nesting area is in eastern Florida, which may be the second-most important worldwide nesting concentration after Oman (Dodd 1988, Ehrhart 1989, NRC 1990, NMFS 1990). The best estimate of annual nesting on southeastern U.S. beaches is just over 14,000 nesting females in an "average year" (Ehrhart 1989), with over 90 % of the activity occurring on eastern Florida beaches. Within this subregion, the greatest density of nesting activity occurs in south Brevard County. Over a seven year period, nest densities averaged 441 clutches per kilometer per year (Redfoot and Ehrhart 1989). Much smaller but important regular nesting aggregations occur in South Carolina, Georgia, and North Carolina (NRC 1990, NMFS 1990).

Population Size

It is difficult to obtain accurate information on population size for any loggerhead population, given the difficulties of finding and following the turtles at sea. This is especially true for the younger age classes that may spend several years adrift in *Sargassum* in the north Atlantic gyre (Carr 1986, 1987, Schwartz 1988). However, it is generally agreed that the most suitable index to population stability results from monitoring the numbers of females nesting at a given rookery from year to year (Ehrhart 1989, NRC 1990, NMFS 1990). The annual

fluctuations in the numbers (or proportions) of reproductively active female turtles (Frazer 1984, 1989, Frazer et al. 1990) make it difficult to assess trends in population size based on numbers of nesting females for any species, and this is recognized even by those who suggest that numbers of nesting females provide the best indicator available (Ehrhart 1989, NRC 1990, NMFS 1990). It is widely recognized that such annual fluctuations may mask general trends in population size for loggerheads unless studies are carried out over several decades (Richardson 1982, Frazer 1983, NRC 1990). Nonetheless, most estimates of population increase, stability, or decline, currently rely on monitoring numbers of nesting females.

Detailed information on population sizes and trends not only is difficult to obtain, but often is even more difficult to interpret once obtained. The problem of assessing worldwide population levels is exacerbated by the fact that few countries support the level of research necessary to monitor long-term fluctuations in sea turtle populations. It is generally believed that there has been a historical decline in the numbers of loggerheads in several areas outside the U.S. (Table 1).

One area for which long-term data are available is on the Bundaberg coast in south Queensland, Australia. The nesting beaches of the Bundaberg coast, as well as those of the islands of the Capricorn Bunker Groups of the southern Great Barrier Reef, have shown a decline in numbers of nesting loggerheads in recent years (Limpus and Reimer 1994). Numbers of nesting loggerheads on the Bundaberg coast have fallen from an average of well over 400 during the late 1970's, to fewer than 300 each year of the late 1980's. In fact, there has been a steady monotonic decline from just under 300 in 1985 to well under 200 in 1989. This general decline in numbers of nesting loggerheads is corroborated by additional studies at the turtles' feeding grounds (Limpus and Reimer 1994).

Long-term studies of nesting females have recently provided information that was not available at the time of the last status review of loggerheads (Mager 1985). Numbers of nesting females in South Carolina declined by over 26 % in a five year period (Hopkins-Murphy and Murphy 1988). Nesting activity on Little Cumberland Island, Georgia during the late 1980's had decreased by 50 % from levels seen there throughout the mid-1970's and was only 25 % of that observed there in the mid-1960's through early 1970's (NRC 1990). Thus, long-term study has documented a decrease of more than 5 % a year in South Carolina over a 5-year period, and of more than 3 % a year in Georgia over a 28-year period. The rate of decline in numbers of nesting loggerhead females in these areas is similar to the rate of decline of the most severely endangered sea turtles, the Kemp's ridley (*Lepidochelys kempii*) in Mexico (Frazer 1986b).

As mentioned earlier, over 90 % of the loggerhead nesting activity in the U.S. occurs in eastern Florida. No similar decline has been demonstrated for the Florida nesting aggregations. However, the National Research Council's (1990) suggestion that the number of loggerheads nesting on Hutchinson Island, Florida might be increasing may be overly optimistic. Their results (NRC 1990) do not indicate any significant trend towards either an increase or a decrease (e.g., $p > 0.06$). Martin et al. (1989), who study loggerheads in the area, also reported no significant ($p > 0.05$) increase or decrease in numbers of females nesting at Hutchinson Island over the 16-year period from 1973 to 1989. In addition, Ernest et al. (1989) reported no significant increase or decrease in 13 years of monitoring captures of loggerheads in the intake canal for the St. Lucie Power Plant on Hutchinson Island, indicating that abundance of loggerheads in the waters off the island has not increased in the past several years. Thus, despite suggestions to the contrary (NRC 1990), there appears to be no indication of any statistically significant increase of loggerheads in the Hutchinson Island area, and it seems best to assume (guardedly) that the numbers have been stable in

recent years rather than increasing. As Martin et al. (1989) suggested, the predicted increased in human traffic on the beach, as well as attempts at erosion control, might negatively impact nesting on Hutchinson Island in the near future.

As mentioned above, south Brevard County supports the highest density of loggerhead nests of any U.S. beach. The linear regression performed on numbers of nests deposited in south Brevard County, Florida, over a seven year period (Table 2) also indicates no significant ($p > 0.30$) increasing or decreasing trend in the numbers of nests recorded by Redfoot and Ehrhart (1989) for the 21 km study area. Although nesting in this extremely important area appears to be relatively stable over the past few years, those who study the loggerheads there urge caution concerning the impact of the anticipated increase in the human population in the area (Redfoot and Ehrhart 1989). They also call attention to the fact that raccoon depredation of loggerhead nests almost tripled in that area between 1985 and 1988 (Redfoot and Ehrhart 1989), indicating that although numbers of nests may be remaining fairly stable, the numbers of hatchlings emerging from those nests may be declining.

Listing Factors

Numerous authors have called attention to the current and impending degradation of loggerhead nesting beaches, particularly in Florida (e.g., Martin et al. 1989, Redfoot and Ehrhart 1989). Such degradation may result from any number of causes, including "nourishment" or augmentation of eroded beaches (Nelson and Dickerson 1989a, NRC 1990, NMFS 1990), beach "armoring" or attempts at erosion control (NRC 1990, NMFS 1990), increased human traffic on the beach (Mortimer 1989), attempts to clean the beaches of debris with heavy equipment (NRC 1990, NMFS 1990), photopollution (Nelson and

Dickerson 1989b, Witherington 1986) or even the unintentional introduction of exotic vegetation onto nesting beaches (LeBuff 1990, Schmelz and Mezich 1988).

Quantitative estimates of human-induced mortality of nesting females, eggs and hatchlings are not readily available. However, during 1988 alone, the Florida Department of Natural Resources received reports of over 10,000 hatchlings that had been disoriented by artificial lighting (NMFS 1988). With the anticipated continued human population growth and development of beach front property in Atlantic and Gulf coast states in general, and in Florida in particular, the degradation of nesting environments seems likely to increase in the immediate future.

Efforts at providing nesting refuges in areas of heavy nesting (Jackson et al. 1988, Possardt and Jackson 1989, Possardt 1991) will doubtlessly help to mitigate some of these problems. However, if the degradation of nesting habitat outside the refuge results in loggerheads increasing the density of their nests inside the refuge, this could result in a degradation of nesting habitat inside the refuge. It is anticipated that there would be an increase in the number of nests destroyed as turtles nesting later may dig up nests that were laid earlier (Bustard and Tognetti 1969). In addition, we are not able to predict the effect on the beach ecosystem should loggerhead nesting decline dramatically in areas outside the refuge. As Conway (1988:265) warned at the National Forum on Biodiversity "...sustaining species...in small fragmented refuges provides little to the Earth in the way of basic ecological services" except in those isolated areas.

Degradation of loggerhead habitat is not restricted to the terrestrial phase of their life cycle on beaches. As Carr (1984:231) reminded us, "Turtle food comes mainly from the bottom in the shallow fringes of the sea..." These near-shore and continental shelf waters, where adult

and large juvenile loggerheads spend much of their time, are also the areas on which human activities potentially have their largest impact. Dredging, sea floor mining, oil spills, industrial and agricultural runoff, and dumping of debris all can potentially degrade the marine habitat occupied by loggerheads (Witzell 1989, NRC 1990). Several studies provide quantitative information on loggerheads insofar as the ingestion of plastics or entanglement in marine debris are concerned (e.g. Stanley et al. 1988, Plotkin and Amos 1988, Plotkin 1989), but quantification of most other aspects of the degradation of the marine habitat is not yet available (Witzell 1989). Nevertheless, it is expected that increased human population pressures will exacerbate the degradation of the near-shore and continental shelf areas of the marine environment and thus negatively impact loggerhead foraging and breeding areas in the immediate future.

There is presently no legal commercial or recreational take of loggerheads in waters under U.S. jurisdiction. Although they may be subject to exploitation in other countries (see Table 3 in Mager 1985), most countries have at least some regulations affecting the take of sea turtles (NRC 1990). The National Research Council's Committee on Sea Turtle Conservation does not consider the illegal take of sea turtles or their eggs to be a significant problem in the United States (NRC 1990).

Dodd (1988), provides extensive review lists of predators, commensals and parasites of loggerheads. Stancyk (1982) earlier provided a review of much of the published information on predators of sea turtles and their eggs. However, little quantitative data are available that would give an accurate indication of the overall effect of disease or depredation on population levels, although depredation is known to be especially heavy on hatchlings and eggs. For example, up to 80% of loggerhead nests may be destroyed by the raccoon, *Procyon lotor*, on some southeastern U.S. beaches (NRC 1990; NMFS 1990) in some years.

Most turtle species display high mortality rates for eggs and young juveniles but low mortality rates for adults and larger juveniles (see review by Iverson 1991a). Hence, heavy depredation of eggs and hatchlings may be the norm for most turtle species, and the selective pressure resulting from such heavy depredation of early life stages is thought to have been responsible for the evolution of longevity and iteroparity in the Order Testudines (Wilbur and Morin 1988, Congdon and Gibbons 1990). The pattern holds up for the all turtle species for which complete life tables have been constructed (e.g., Tinkle et al. 1981, Crouse et al. 1987, Frazer et al. 1990, Iverson 1991b). Thus, it is difficult to ascertain whether occasional years of heavy depredation of loggerhead eggs and hatchlings is a normal or abnormal occurrence in the life of the species in any particular area. It is clear, however, that sustained levels of heavy predation on these early life stages can severely threaten loggerhead populations if adults and larger juveniles are not experiencing their typically high natural survival rates due to human-induced mortality in shrimp trawl nets (Crouse et al. 1987).

Although Mager (1985) suggested in the last status review that existing regulatory mechanisms were sufficient for the protection of loggerheads, the existing regulations have not prevented the decline of nesting aggregations in some areas such as South Carolina and Georgia (NRC 1990, Hopkins-Murphy and Murphy 1988, Frazer 1986b). Even where numbers of nesting females are relatively constant, there is some question as to whether existing regulations will prevent the continued degradation of beach and near-shore marine habitats (Martin et al. 1989, Redfoot and Ehrhart 1989).

Recent demographic models of loggerhead population dynamics indicate that protection of large juveniles and adults may be the most critical management need in the immediate future, and it is these very life stages that are at greatest risk of capture in shrimp trawl nets (Crouse et al. 1987, Frazer 1989b, NRC 1990). In areas where numbers of nesting females have

been declining, there is some evidence that the use of turtle excluder devices (TEDs) on shrimp trawl nets has resulted in a decrease of stranded loggerheads. Crowder et al. (1994a) reported an overall trend which showed loggerhead strandings declining at about 5 - 6 % per year. However, given the resistance of the shrimping industry, confusion and equivocation in terms of the TED regulations in some areas earlier (Rudloe and Rudloe 1989, Williams 1990), and recent increases in sea turtle strandings in the Gulf of Mexico (Shaver 1994), presumably a result of incorrect installation of TEDs in some shrimp trawl nets, it remains to be seen whether enforcement can ensure that TEDs will be deployed consistently and effectively. Early estimates of annual survival rates for adult females (0.81) and larger juveniles (0.70) were obtained for turtles in U.S. Atlantic waters before the introduction of TEDs (Frazer 1983, 1987). There is a great need for similar estimates of annual survival rates of large juvenile and adult loggerheads in these waters once an effective history has been established over several years with TEDs in place, as well as for the Gulf of Mexico. A recent model of loggerhead population dynamics (Crowder et al. 1994b) has shown that it may take up to 70 years or more before the deployment of TEDs on shrimp trawls results in any substantial observable increase in the numbers of nesting females.

Henwood and Stuntz (1987) estimated that between 9,000 and 10,000 loggerheads were killed each year in U.S. Atlantic and Gulf of Mexico waters as a result of drowning in shrimp trawl nets. The National Research Council (1990) provided convincing arguments that these previous estimates might be low. Their own order of magnitude numerical estimates of human-induced mortality of juvenile and adult sea turtles in U.S waters (Table 3) indicate that incidental capture in fishing gear is the most important known source of human-induced mortality (NRC 1990), and may be as much as five times the level estimated by Henwood and Stuntz (1987). Inundation of eggs by heavy rains or tides and erosion or accretion of sand on nesting beaches are among the most important abiotic factors affecting

survival of eggs and hatchlings. However, none of these is considered to be an important source of mortality threatening loggerhead populations (NRC 1990, NMFS 1990).

Conclusions

In the last status review of loggerheads, Mager (1985) concluded that there was inadequate information available to assess whether its status had changed since the initial listing as threatened wherever it occurs (1978). Subsequent information presented above clearly demonstrates that its status has changed for the worse in recent years. There can be little doubt that numbers of nesting females have declined in Georgia and South Carolina (Frazer 1986b, Hopkins-Murphy and Murphy 1988, NRC 1990), while Florida's nesting aggregations have held fairly constant (Table 2, Martin et al. 1989, NRC 1990). Increased human presence in areas of heavy nesting is likely to impact dense nesting populations in Florida in the near future (Redfoot and Ehrhart 1989, Martin et al. 1989), even if nesting beach refuges are provided. There certainly can be no case for delisting loggerheads, given the evidence presented herein. Therefore, recommendation is that loggerheads retain the listing as threatened wherever they occur. Although the current recommendation is to retain the "threatened" designation, further study is needed before the next status review concerning U.S. loggerhead population stock structure. Stoneburner (1980) presented morphological data suggesting possible differences in carapace shape between loggerheads nesting in Georgia and those nesting in Florida. Recent biochemical evidence indicates that the adult loggerheads nesting in Georgia may represent a population that is distinct from those turtles nesting in Florida (Bowen et al. 1993), despite earlier findings to the contrary (Smith et al. 1977). The possibility that loggerheads along the southeast Atlantic coast represent at least two distinct populations deserves close attention and further research. If there are, indeed, two separate demes, then the northern one (i.e., those nesting in GA, SC, NC, and

northward) is in great difficulty. Its calculated rate of decline is as great or greater than that of the Kemp's ridley (Frazer 1986b, Hopkins-Murphy and Murphy 1988). Although the Florida nesting aggregation has been stabilized in numbers over the past several years, it is threatened with additional detrimental effects because of an increase in the coastal human population. Should additional research confirm the existence of the two demes, and should the decline of the northern nesting population continue its decline, careful consideration must then be given to listing them as endangered when their status is next reviewed.

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Table 1. Areas for which there has been a historical decline in numbers of loggerhead sea turtles, *Caretta caretta*, or for which there is a suspected decline due to heavy disturbance or exploitation (Demetropoulis and Hadjichristophoru 1989, Ehrhart 1989, Gil and Leon 1988, Groombridge 1982, Groombridge and Whitmore 1989, Hewavisenthi 1990, Ross 1982, Veniselos 1986).

<u>Evidence of Historical Decline</u>	<u>Suspected Decline</u>
Bahamas	Azores
Colombia	Cape Verde Islands
Cuba	Dominican Republic
Cyprus	Guatemala
Greece	Italy
Honduras	Madagascar
Israel	Madeira
Japan	Mozambique
Mexico	West Africa
Panama	
San Andreas Archipelago	
Sri Lanka	

Table 2. Number of nests deposited annually in a 21 km study area of the beach in south Brevard County, Florida by loggerhead sea turtles, *Caretta caretta* (Redfoot and Ehrhart 1989, Owen et al. 1991, Owen et al. 1993, Owen et al. 1994).

<u>Year</u>	<u>Number of Nests</u>
1982	7,995
1983	9,423
1984	7,753
1985	10,240
1986	10,745
1987	9,780
1988	8,838
1989	9,381
1990	14,328
1991	13,194
1992	12,754
1993	10,591

Table 3. Estimated annual numbers of deaths and relative importance of sources of human-induced mortality of juvenile and adult loggerhead sea turtles (*Caretta caretta*) in U.S. waters. Relative Importance: 1 = high; 2 = moderate; 3 = low; 4 = unimportant. (Adapted from NRC 1990).

<u>Cause</u>	<u>Mortalities/year</u>	<u>Relative Importance</u>
Shrimp Trawling	5,000-50,000	1.0
Other Fisheries	500-5,000	2.5
Beach Development	?	3.0
Dredging	50-500	3.0
Entanglement	?	3.0
Oil Platform Removal	10-100	3.0
Collisions with Boats	50-500	3.0
Directed Take	5-50	3.0
Power Plant Entrainment	5-50	3.0
Recreational Fishing	?	3.0
Beach Vehicles	?	3.0
Beach Lighting	?	3.5
Beach Replenishment	?	4.0
Toxins	?	?
Ingestion of Plastics or Other Debris	?	?

East Pacific Green Sea Turtle, *Chelonia mydas*

Prepared by Javier Alvarado and Alfredo Figueroa

Biological Background

Historically the black sea turtle has also been known as the East Pacific green turtle in English, and as tortuga negra, tortuga prieta, caguama prieta, tora and sacasillo in Spanish (Cliffon et al. 1982, Cornelius 1986). The East Pacific green turtle and the green turtle (*Chelonia mydas*) share the following features that distinguish them from other sea turtles: four pairs of lateral scutes on the carapace; one pair of elongated prefrontal scales between shell scutes that do not overlap; and the tomium of the lower jaw with a sharply serrated cutting rim that corresponds with strong ridges on the inner surface of the upper tomium (Carr 1952, Marquez 1990).

In 1868, Bocourt proposed the binomial form *Chelonia agassizii* based on the examination of East Pacific green turtle specimens from the Pacific coast of Guatemala. The usage of this binomial, however did not gain general acceptance and in the present century, the East Pacific green turtle often has been referred to as *Chelonia mydas agassizii* even though there has not been a detailed published argument to reduce it to subspecific rank. There has been recent controversy over the taxonomic status of the East Pacific green turtle and whether or not it is distinct from the Pacific green turtle *Chelonia mydas*. In distribution the East Pacific green turtle is restricted to the east Pacific, where according to Pritchard (1983) there exists occasional sympatry between black and green turtles without apparent intergradation. Morphological differences (Cornelius 1986, Groombridge and Luxmoore 1989, Figueroa and Alvarado 1990) and osteological features (Kamezaki and Matusi 1995) show a distinction between *Chelonia agassizii* and other *Chelonia*. The East Pacific green turtle is distinguished from the green turtle mainly by size, coloration and carapace shape. The carapace of adult East Pacific green is narrower, more strongly vaulted and more indented over the rear flippers than in the green turtle (Cornelius 1986, Marquez 1990). In size and weight the East Pacific green is conspicuously smaller and lighter than the green turtle. In the rookeries of Mexico, the mean size for nesting East Pacific green is 77.0 cm in straight carapace length (SCL) (Alvarado and Figueroa 1989a). On the Galapagos Islands, the mean SCL for nesting East Pacific green is 75.0 cm (Marquez 1990). Adult female East Pacific greens weigh

between 65 - 125 kilograms (Cornelius 1986). On the other hand, the typical adult female green turtle has an average shell length of 96.6 cm (SCL) and weighs between 65 - 160 kilograms (Groombridge 1982). East Pacific green males in the rookeries of Mexico are smaller than females with an average SCL of 75.0 cm (Figueroa 1989). The mean size for East Pacific green hatchlings is 4.5 cm in SCL (Zamora 1990). Morphological data are supported by nuclear DNA analysis which showed that Pacific Mexico and Galapagos Islands *Chelonia* were distinct from other *Chelonia* populations (Karl et al. 1992). However, mtDNA analyses do not show distinctness of the Eastern Pacific *Chelonia* from *Chelonia* populations in other regions of the world (Bowen et al. 1992, Dutton et al. In press).

From a dorsal view, true color of adult East Pacific green turtle is characteristically dark. The carapace and upper surfaces of the head and flippers is slate grey to black whereas the plastron varies from bluish to dark grey (Caldwell 1962). Hatchlings are black to dark gray above and white below with a white border around the dorsal edge of the carapace and flippers (Cornelius 1986) while young individuals are usually brightly colored with a mottled or radiating pattern of brown, olive or yellow (Caldwell 1962).

There is no sexual dimorphism in subadult male and female turtles, however, adult males have longer tails that reach well beyond the rear edge of the shell, while adult females have shorter tails that barely reach beyond the rear edge. Mature males have recurved long claws for grasping the female and their plastrons become softened during breeding seasons. Carapace length at sexual maturity in females is about 55.0 cm (SCL) (Alvarado and Figueroa 1989a). An estimate of age at sexual maturity for the Galapagos population is > 46 years (MacFarland 1984).

Adult East Pacific green turtles are primarily herbivorous, eating sea grasses and algae (Mortimer 1982) and in some areas they also feed on shellfish, jellyfish and other marine organisms (Brown and Brown 1982, Cornelius 1986). Predators of adult sea turtles are primarily man and sharks (Stancyk 1982). Eggs are eaten by skunks, boars, domestic and feral dogs, pigs, coatis, ghost crabs, ants, a beetle (*Trox suberosus*) and a scavenger fly (Sarcophagidre) (Green and Ortiz-Crespo 1982, Alvarado and Figueroa 1985, Marquez 1990). Hatchlings are eaten by coatis, shorebirds, lizards, ghost crabs, vultures, bonyfishes and sharks (Marquez 1990).

East Pacific green turtles occur along the west coast of America from central Baja California and the Gulf of California (Alvarado and Figueroa 1990a) to southern Peru (Marquez 1990). Small numbers or isolated individuals have been reported from California (Dutton and McDonald 1990), British Columbia, Canada and Chile (Marquez 1990). The main nesting sites are two neighboring beaches at Maruata Bay and Colola on the coast of Michoacan, Mexico (Cliffon et al. 1982), and on the Galapagos Islands in Central America (Hurtado 1984). There are also less important nesting grounds in Mexico, namely the Pacific coast of Guerrero, Jalisco, Oaxaca and Chiapas, and the islands of Clarion and Socorro (Marquez 1990) and along the Central American Pacific coastline (Cornelius 1982).

The nesting season varies with location. Nesting occurs in Michoacan between August and January, with a peak in October-November (Alvarado and Figueroa 1985), between March and July at the Socorro and Clarion Islands (Marquez 1990), and between December and May with a peak in February on the Galapagos Islands (Green and Ortiz-Crespo 1982). Nesting cycles show differences between northern and southern populations. In Michoacan females typically nest in two or three year cycles and deposit between one and seven clutches per season at about 12-14 day intervals (Alvarado and Figueroa 1990a). In the South, on the Galapagos Islands, females nest in two to five year cycles and deposit between one to five clutches per season at about 14 day intervals (Marquez 1990). Average number of eggs per clutch varies between 65 in Michoacan (Alvarado and Figueroa 1990a) to 80 in the Galapagos Islands (MacFarland 1984). After 42 to 62 days of incubation (Marquez 1990) hatchlings emerge mostly at night and travel quickly to the sea. After some time in a pelagic existence, they enter shallow inshore and nearshore waters as juveniles.

Population Size

The status of sea turtle species is perhaps best indicated by long-term changes or trends in the sizes of individual populations. Because females repeatedly return to the same beaches to nest and because this is the time in their life cycle in which they are most available for direct counting, counts of nesting females or nests provide the best available long-term index of the status of populations.

The Michoacan nesting population consists of about one-third of the entire mainland-nesting East Pacific green turtle population in the east Pacific (Cliffon 1982). Nesting females in

Michoacan have exhibited a clear decline in numbers over the last 40 years. In the early 1970's it was estimated that about 25,000 female turtles nested in Michoacan (Cliffton et al. 1982). Earlier observations tend to corroborate this figure; Peters (1962) reported tracks of around 250 turtles on a 0.8 km of beach at Maruata Bay in August 1950, some two months before peak nesting. Cliffton et al. (1982) take this to indicate about 900 turtles nesting in Maruata Bay within several days of Peters' observations. This figure is 50 percent higher than the number estimated nesting in the bay for the entire 1979 season. By the middle 1970s nesting turtles were no longer common in August. In the past 10 years the estimated nesting population in Michoacan has ranged from a high of 5,586 females in 1982 to a low of 940 in 1984 (Alvarado and Figueroa 1990a). Recent estimates show that the number of East Pacific green turtles in the Michoacan nesting population is now extremely low. An estimated 470 female turtles nested in the 1994-1995 season (Alvarado et al. 1995). In the Galapagos Islands, East Pacific green turtles occur in moderate numbers. Between 1,200 - 3,250 females nest in the archipelago annually (Green 1994).

Although traditionally used as a food source by many coastal inhabitants, the primary cause of decline in East Pacific green turtle numbers has been the systematic exploitation of eggs and adults for marketing. Harvest of adults and eggs in Pacific Mexico increased steadily in the present century, especially in the 1960's and 1970's (Alvarado and Figueroa 1990a). Although largely undocumented, incidental catch by shrimp trawlers is probably another major mortality factor in Mexico (Groombridge 1982) as well as in Central America (Cornelius 1982).

Listing Factors

1. The Present or Threatened Destruction, Modification or Curtailment of its Habitat or Range.

Since 1980, the East Pacific turtle nesting beaches in Colola and Maruata, Michoacan, Mexico, have been protected by armed patrols. Since 1986, they have been declared Natural Reserves, and have been protected from development (Alvarado and Figueroa 1989b). The Galapagos Islands are a National Park and all nesting beaches are completely protected (Groombridge 1982). Accordingly the main nesting beaches are not yet threatened by destruction or modification. The many

smaller nesting beaches in Mexico and Central America are largely unprotected from habitat destruction.

In its oceanic environment East Pacific green turtle should be adversely affected by habitat alterations and substances to the same degree as other sea turtles. The following list includes some of the activities and substances that may adversely affect or degrade the habitat of these turtles.

- A. Pollutants from industrial and residential development. Discharges of agricultural and industrial chemicals, petroleum products, and domestic sewage may have indirect effects on herbivorous turtles by reducing food sources through degradation of grass pastures, and direct effects such as a possible reduction in health and fitness of individual animals that may be manifested by mortality and the disruption of physiological functions (Coston-Clements and Hoss 1983). However, levels of pollutant discharges for much of the east Pacific are unknown and the direct effects of these pollutants on East Pacific green turtles have not yet been investigated.
- B. Disposal of garbage at sea. The growing presence of trash in the marine habitat is an increasing threat to endangered populations of sea turtles and other marine life. Marine debris poses a variety of threats to sea turtles from ingestion to entanglement. Carr (1987) pointed out that sea turtles of all species eat plastic scraps and other buoyant material and become entangled in lines and netting discarded by fishermen. Plastic can become impacted in an animal's digestive system, causing problems ranging from reduced nutrient absorption, intestinal damage, or false feeling of satiation to blockages resulting in death from starvation (Balazs 1985). Turtles entangled in lines or netting may be subjected to injury, strangulation, starvation, and drowning (Carr 1987).

Although traditionally used as a food source by many coastal inhabitants, the primary cause of decline in East Pacific green turtle numbers has been the systematic exploitation of eggs and adults for marketing. Harvest of adults and eggs in Pacific Mexico increased steadily in the present century, especially in the 1960's and 1970's (Alvarado and Figueroa 1990a). Although largely

undocumented, incidental catch by shrimp trawlers is probably another major mortality factor in Mexico (Groombridge 1982) as well as in Central America (Cornelius 1982).

- C. Power boats. Power boats can injure or kill species of sea turtles (Coston-Clements and Hoss 1983). During 1989 in San Diego Bay, several East Pacific green turtles were found dead by injuries caused by power boats (P. Dutton, pers. comm.).

2. Overutilization for Commercial, Recreational, Scientific and Educational Purposes.

The use of East Pacific green turtles for scientific and educational purposes, while unquantified, is undoubtedly little and not a contributing factor in the decline of these populations. In Mexico, the scientific take of East Pacific green turtles is controlled by a permit program designed to protect the species.

Excessive exploitation is the overwhelming threat to East Pacific green turtle populations throughout most of their distribution range, and has evidently caused dramatic declines in numbers over the present century.

Prior to the 1950s the coastal breeding sites of the East Pacific green turtle in Michoacan were relatively undisturbed as the coastline was virtually uninhabited and of difficult access. During the 1950s coastal areas were increasingly cleared and the coastal settlements of Maruata and Colola were established. Access to the area remained difficult by land and, at that time, use of turtles by the local Nahuatl Indians was at a low-subsistence level (Alvarado and Figueroa 1990b). During the late 1960s commercial markets for sea turtle products opened up in the area, involving both leather and eggs. During the early 1970s about 70,000 eggs were collected each night of the breeding season at Colola and a further 10,000 - 20,000 from Maruata Bay (Cliffon et al. 1982). This total harvest of eggs continued until 1980 when armed protection of Colola and Maruata was begun and hatcheries for the relocation of nests were established (Alvarado and Figueroa 1985).

Until the end of 1963 processing of turtle leather in Mexico was on a local, artisanal level (Groombridge and Luxmoore 1989). During 1964 with the increase in international demand for the product, larger scale, industrial processing began (Alvarado and Figueroa 1989b). From the mid-1960s to mid-1970s East Pacific green turtles in Michoacan were harvested mainly for their skins (Alvarado and Figueroa 1989b). From 1965 to 1977 about 165,000 turtles were harvested in the Mexican Pacific (Groombridge and Luxmoore 1989). In 1973 a new coastal highway reached the area and smugglers from the northern states of Sinaloa and Sonora gained easy access to the Michoacan turtles and since then most poached turtles have been smuggled to the north for meat consumption (Alvarado and Figueroa 1990a).

Cliffon et al. (1982) note that historically the East Pacific green turtle was abundant within the Gulf of California and along the Pacific coast of Baja California. Large-scale commercial exploitation of turtles in the northern Mexican feeding grounds began in the 19th century, considerably earlier than at the Michoacan nesting grounds. At the turn of the century an estimated 1,000 live turtles a month were sent to San Diego from Laguna Ojo de Liebre, Baja California (O'Donnell 1974). From 1956 to 1963 East Pacific green turtles harvested at the northern feeding grounds were the most important component of the Mexican turtle fishery with a live weight production of 3,430 metric tonnes (Groombridge and Luxmoore 1989). Cliffon et al. (1982) noted that by the late 1970s, the remaining sea turtles had been virtually extirpated from the Gulf of California.

In the early 1970s large numbers of overwintering East Pacific green turtles were discovered off the south shore of Tiburon Island in the Gulf of California off the Sonoran coast (Felger et al. 1976). Exploitation of these turtles began in 1975 and by 1980 they had been drastically reduced (Cliffon et al. 1982).

Overall it is certain that since the late 1950s many hundreds of thousands of East Pacific green turtles, quite probably over one million, have been landed on the Pacific coast of Mexico (Groombridge and Luxmoore 1989). This fishery has been by far the most important factor in the collapse of East Pacific green turtle populations.

In Central America as well as in the feeding grounds in Colombia and Panama the East Pacific green turtle is captured as bycatch of the olive ridley fishery (Marquez 1990). In 1982, Brown and Brown reported the existence of a East Pacific green turtle fishery in Peru with an estimated annual catch of about 1,500 - 2,000 individuals, mainly subadults. The current situation of this fishery in Peru is unknown.

3. Disease or Predation.

Cutaneous fibropapillomatosis has been recorded in several green turtle populations (Balazs 1986). These tumors can indirectly cause mortality. Turtles whose vision is blocked by tumors are unable to feed normally and turtles with fibropapillomatosis are more prone to entanglement in monofilament line and other debris (Balazs 1986). There have not been any reports of this disease in East Pacific green turtles from Mexico, Middle America or South America. However, McDonald and Dutton (1990) found fibropapillomas in several juvenile East Pacific green turtles from San Diego Bay, California.

4. Inadequacy of Existing Regulatory Mechanisms.

Taking of sea turtle eggs was prohibited by law in Mexico in 1966 (Marquez *et al.* 1982) and the capture of East Pacific green turtles was prohibited in 1984 (Alvarado and Figueroa 1990b). However, laws are often ignored and poaching of both adults and eggs has been common practice. The establishment of turtle camps at the East Pacific green turtle nesting grounds in Michoacan since 1980 has significantly diminished the poaching of eggs and breeding adults (Alvarado and Figueroa 1990a). In 1990 a presidential decree prohibited capture of all sea turtles and trade in sea turtle products (Aridjis 1990). This should make clandestine trade in turtle products much more difficult in the future.

In Central and South American countries where East Pacific green turtles breed or forage, regulatory mechanisms exist for the protection of sea turtles. As in Mexico, however, laws are often ignored and there is a brisk commerce of eggs in those countries where nesting is common (Cornelius 1982). In the Galapagos Islands only

local inhabitants are legally allowed to fish for turtles and only on a subsistence basis and egg poaching is practically nonexistent (Green and Ortiz-Crespo 1982).

5. Other Natural or Manmade Factors Affecting its Continued Existence.

The effects of natural phenomena on the continued existence of the East Pacific green turtle are not known. However, natural forces that affect the production of eggs and hatchlings, include El Nino, storms, rain and wave surge. These forces can produce beach erosion or accretion, prevent turtles from nesting and destroy eggs and hatchlings.

East Pacific green turtles are incidentally taken by shrimp trawlers as well as in pelagic fisheries such as the longline fishery in the eastern Pacific Ocean. However, the extent of this incidental take and the effect on the population remains unknown (Cornelius 1982, Groombridge 1982). With the population as depauperate as it is, any level of take of large juveniles or adults should be considered a problem for the species.

Conclusion

Excessive exploitation throughout its distributional range has caused a dramatic decline in numbers of the East Pacific green turtle in the present century. Efforts to recover the species must concentrate on enforcing national and international regulatory mechanisms so that nests, hatchlings, adults, subadults and juveniles are given effective protection in both breeding and foraging habitats.

Utilization of adequate gear to reduce the number of East Pacific green turtles caught incidentally in other fisheries, particularly the shrimp trawl fishery, should be required throughout the East Pacific green turtle distributional range. Effective public education to promote conservation of the East Pacific green turtle in coastal communities should be undertaken.

The best available information indicates that the East Pacific green turtle is severely depleted and in danger of extinction. Therefore, this species should be listed as an endangered species throughout its range.

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Leatherback Sea Turtle, *Dermochelys coriacea*

Prepared by Karen L. Eckert, Ph.D.

Biological Background

The leatherback is the largest living turtle and is so distinctive that it is placed in a separate taxonomic family, the Dermochelyidae. Whereas other sea turtles have bony plates covered with horny scutes on the carapace, the slightly flexible carapace of the leatherback is distinguished by a rubber-like texture. The carapace is about four cm thick and is largely constituted of tough, oil-saturated connective tissue raised into seven prominent ridges and tapered to a blunt point posteriorly. A nearly continuous layer of small dermal bones lies just below the leathery outer skin of the carapace. The narrow ribs lack pleural flanges and remain widely separated throughout life. No sharp angle is formed between the carapace and the plastron, resulting in the animal being somewhat barrel-shaped. Adult males are distinguished by a long, thick tail that extends well beyond the posterior carapace margin.

The epidermis is black, with varying degrees of pale spotting, and is scaleless. The underside is mottled pinkish-white and black; the proportion of light to dark pigment is variable. The front flippers are proportionally longer than in other sea turtles and may span 270 cm in an adult. Hatchlings are likewise predominately black with mottled undersides, but differ in being covered with tiny polygonal or bead-like scales. The flippers have white margins, and rows of white scales appear as stripes along the length of the back. Front and rear flippers lack claws. In both adults and hatchlings, the upper jaw bears two tooth-like projections, each flanked by deep cusps, at the premaxillary-maxillary sutures. For a

detailed discussion of anatomy, including embryonic development, see Deraniyagala (1932, 1936), Dunlap (1955), Pritchard (1971), Pritchard and Trebbau (1984), and Renous et al. (1989).

The population of leatherback turtles nesting at the Sandy Point National Wildlife Refuge (NWR) on St. Croix, U. S. Virgin Islands (USVI), has been intensively studied since 1981 under the auspices of the USVI Division of Fish and Wildlife. A similar project was initiated on Resaca and Brava beaches (Culebra Island, Puerto Rico) in 1984 by the U. S. Fish and Wildlife Service. The average curved carapace length (CCL) of nesting females on these beaches is 155 cm (range 137-176 cm) and weights of 262-506 kg have been recorded (Eckert and Eckert 1983, 1985, Basford et al. 1986, 1988, Brandner et al. 1987, 1990, Tucker 1988, Tallevast et al. 1990). Hatchlings average 61.3 mm (n= 398) straightline carapace length and 45.8 g (n = 282) in weight (Eckert et al. 1984). Similar data are reported from Australia (Limpus and McLachlan 1979), French Guiana (Fretey and Girondot 1989), Costa Rica (Hirth and Ogren 1987), and Trinidad (Bacon 1970). Nesting females are larger and smaller, respectively, in South Africa (averaging 164 cm CCL; Hughes et al. 1967) and Pacific Mexico (averaging 144 cm CCL; Sarti et al. 1987). The largest leatherback on record is a male that stranded on the coast of Wales in 1988 and weighed 916 kg (Morgan 1989).

Leatherbacks have the most extensive range of any reptile (71°N to 47°S; Pritchard and Trebbau 1984) and exhibit broad thermal tolerances. The species is commonly reported in New England waters and northward into eastern Canada, as well as occasionally sighted in British Columbia and northward into Alaska. In Trinity Bay, Newfoundland, an adult was observed by fishermen throughout an entire day swimming in open water amongst ice (Goff and Lien 1988). The core body temperature for adults in cold water has been shown to be

several degrees C above ambient (Frair et al. 1972). This may be due to several features, including the thermal inertia of a large body mass, an insulating layer of subepidermal fat, counter-current heat exchangers in the flippers, and potentially heat-generating brown adipose tissue (Mrosovsky and Pritchard 1971, Frair et al. 1972, Greer et al. 1973, Neill and Stevens 1974, Goff and Stenson 1988). The skeleton remains extensively cartilaginous, even in adult animals, and the species is unique among turtles in showing an extensive cartilage canal vascular system in the long bones (Rhodin et al. 1981).

With the exception of very preliminary data on growth in mature females (nesting females returning to Sandy Point NWR gain, on average, about 0.5 cm CCL per year; R. Boulon, unpubl. data), information concerning the growth rate of wild individuals is entirely lacking. Based on the unusual features of the leatherback skeletal system, as well as evidence that some individuals have grown quite rapidly in captivity, Rhodin (1985) has speculated that leatherbacks may reach sexual maturity in 2-3 years. Bels et al. (1988) challenge this hypothesis in their report of a healthy captive leatherback 1200 days (3.28 yrs) old with a carapace length of 82 cm and a weight of 28.5 kg, dimensions which are an order of magnitude smaller than mature leatherbacks measured in the wild. Leatherbacks may indeed reach sexual maturity at an earlier age than other sea turtles; however, information on the growth of juveniles in the wild is necessary before growth rates can be accurately predicted.

Analysis of stomach contents has shown that leatherbacks feed primarily on cnidarians (medusae, siphonophores) and tunicates (salps, pyrosomas) in temperate and boreal latitudes (Bleakney 1965, Brongersma 1969, Den Hartog and Van Nierop 1984, Davenport and Balazs 1991). The turtles are sometimes observed in association with jellyfish, such as *Stomolophus* (Leary 1957, Lohoefer et al. 1988), *Aurelia* or *Chrysaora* (Musick 1988, Collard 1990), but actual feeding behavior only occasionally has been documented. Foraging has been

observed at the surface (e.g., on *Aurelia*, as reported by Eisenberg and Frazier 1983), but is also likely to occur at depth. Based on offshore studies of diving by adult females nesting on St. Croix, Eckert et al. (1989a) proposed that the observed inter-nesting dive behavior reflected nocturnal feeding on vertically migrating zooplankton, chiefly siphonophore and salp colonies. Den Hartog (1980) speculated that foraging may occur at depth after finding nematocysts from deep water siphonophores in leatherback stomach samples.

The evidence currently available from tag returns and strandings in the western Atlantic suggests that adults engage in routine migrations between temperate and tropical waters, presumably to optimize both foraging and nesting opportunities. Leatherbacks tagged while nesting in the Caribbean sometimes strand on New England shores (Pritchard 1973, 1976, Lambie 1983, Boulon et al. 1988). Rhodin and Schoelkopf (1982) reported a gravid female stranded in August on the shores of New Jersey, the eggs possibly residual from her last clutch of the season in more southerly latitudes. Conversely, an adult female tagged in Chesapeake Bay in May 1985 was killed in the Gulf of Guacanayabo, Cuba, in July 1986 (Keinath and Musick 1990). The species composition of epibiotic barnacle communities on Caribbean-nesting leatherback provides indirect evidence that gravid females embark from and subsequently return to temperate latitudes (Eckert and Eckert 1988). The longest documented movement is that of an adult female who traveled 5,900 km to Ghana, West Africa, after nesting in Surinam (Pritchard 1973). Recent studies by Morreale et al. (1993) using satellite telemetry to monitor post-nesting movements of leatherbacks from the Pacific and Caribbean coasts of Costa Rica have found that nesting cohorts travel along similar migration routes and that these routes tend to be located along deepwater bathymetric contours.

Long distance (> 100 km) intra-seasonal movement sometimes occurs among nesting beaches; for example, between French Guiana and Surinam (Pritchard 1973) and between St. Croix and Puerto Rico (Eckert et al. 1989b). Keinath and Musick (1993) used satellite telemetry to monitor the inter-nesting movements of a female leatherback tagged while nesting on St. Croix. After leaving St. Croix, this leatherback traveled a minimum distance of 515.2 km and nested twice more on Vieques and Culebra Islands, Puerto Rico.

An observation of courtship and/or mating in Puerto Rican waters was described by Carr and Carr (1986); however, there is recent evidence that mating typically occurs prior to (or during) migration to the nesting ground (Eckert and Eckert 1988). Nesting is generally nocturnal. Preferred nesting beaches have deep and unobstructed offshore access (Hirth 1980, Mrosovsky 1983). When nesting occurs on high energy, spatially unpredictable beaches, it is not uncommon for large numbers of eggs to be lost to erosion (Mrosovsky 1983, Eckert 1987). The most detailed reproductive data available for entire nesting colonies have been collected at Sandy Point NWR, St. Croix, and on Culebra Island, Puerto Rico. In these areas, nesting commences in March (rarely in February) and continues into July. Females renest on average every 9-10 days, deposit an average of 5-7 nests per annum (observed range 1-11), and remigrate at 2-3+ year intervals. Annual project reports record that clutch size averages 116 eggs, including 80 yolked eggs, on Sandy Point (1982-1989) and 103 eggs, including 70 yolked eggs, on Culebra Island (1984-1989). Similar data have recently been reported elsewhere in the Western Atlantic; for example, in French Guiana (Fretey and Girondot 1989) and Costa Rica (Hirth and Ogren 1987). Data from Culebra suggest that remigrants (returning turtles tagged during a previous nesting year) lay more clutches per annum than do turtles recorded nesting at this rookery for the first time (Tucker and Frazer 1991).

The temperature in the nest during the 55-75 days of incubation influences the sex of hatchlings. The "pivotal temperature" (= 1:1 sex ratio) for leatherback eggs is estimated to be 29.25°C-29.50°C in Surinam and French Guiana (Mrosovsky et al. 1984, Rimblot-Baly et al. 1986-1987). Pivotal temperatures have not been defined elsewhere for this species. Several authors have cautioned against artificial incubation techniques which potentially bias sex ratios (e.g., Mrosovsky and Yntema 1980, Morreale et al. 1982, Dutton et al. 1985). Under natural circumstances hatchlings emerge from the nest in the early evening and, to the best of our knowledge, orient toward the ocean using light; that is, using the brightness of the seaward horizon as their primary cue. Once in the water there is some evidence that hatchlings maintain their orientation by swimming at a fixed angle relative to waves (Lohmann et al. 1990). Locomotion is described by Davenport (1987). Nothing is known of the behavior or survivorship of post-hatchlings. In his review of the pelagic stage of post-hatchling sea turtle development, Carr (1987) found no evidence that young leatherbacks associate with *Sargassum* or epipelagic debris. On 19 May 1989, an injured juvenile (29 cm CCL) was found in the surf off Playa Humacao, Puerto Rico. It was transferred to Coral World, St. Thomas, for rehabilitation and subsequently died there (Johnson 1989). Beyond a few such chance encounters, there are no data on immature leatherbacks under U. S. jurisdiction.

Population Size

Nesting grounds are distributed circumglobally (ca. 40°N to 35°S), with the Pacific coast of Mexico supporting the world's largest known colony of nesting leatherbacks. The only available estimates of the world population of leatherback turtles are confined to breeding females. From nest counts and a knowledge of intra- and inter-annual nesting periodicities, the number of breeding females can be approximated. Pritchard (1982) estimated the number

of adult female leatherbacks worldwide to be 115,000, with about half of them nesting along the Pacific coast of Mexico. Recent declines in the number of leatherbacks nesting in Mexico and elsewhere have occurred and Spotila et al. (In press) estimate that only 20,000 to 30,000 female leatherbacks exist worldwide. During the 1986-1987 breeding season, an estimated 4796 nests were laid on 4.5 km of beach at Mexiquillo, Michoacan, Mexico (Sarti et al. 1989). Nesting at this site has been steadily dropping, however. An estimated 1200 nests were laid in 1990-1991, and a mere 70 nests were laid in 1993-1994 (Laura Sarti M., UNAM, unpubl. data). It is not uncertain whether the decline reflects a natural fluctuation (perhaps related to El Nino), or is a warning that the population is in serious jeopardy. The other large colony in the western hemisphere is Yalimapo-Les Hattes, French Guiana, where the total number of adult females is estimated at 14,700-15,300 (Fretey and Girondot 1989). As erosion has degraded nesting beaches in French Guiana, the large colony there has spilled over into Surinam. There were fewer than 100 leatherback nests laid in Surinam in 1967, but annual numbers have risen (amid natural year-to-year fluctuation) to 5565 nests in 1977 and 9816 nests in 1987 (Reichart and Fretey 1993).

Until recently, large colonies also flourished in the eastern hemisphere. At least 13,000 leatherback nests were reported in 1984 on 17.8 km of coast extending eastward from Tanjung Jamursba on the Bird's Head Peninsula, Irian Jaya, making this Indonesian site one of the most important in the world (Bhaskar 1985). In 1991, Betz and Welch (1992) visited the Jamursba-Medi rookery and reported that nesting levels had declined to a mere 25% of those reported by Bhaskar (1985); the near total collection of eggs has most certainly contributed to the population's demise. Similarly, the beach at Rantau Abang (Terengganu, Malaysia), once considered a major nesting area, has experienced a dramatic decline as a result of intensive egg collection. The data show a steady drop in nesting activity from 6721 landings in 1968 to 372 in 1988 (Chua 1988) to 207 in 1991 (Sukarno Wagiman, Marine

Fisheries Resource Research Center, Terengganu, in litt., December 1991). In addition, about 400 leatherbacks are caught annually off the coast of Terengganu (Brahim bin Salleh et al. 1987); trawl nets are responsible for some 60% of the total number of turtles caught, with the balance attributed to drift nets (Chan et al 1988). Population declines also have been reported in India, Sri Lanka, and Thailand (see Ross 1982).

Pritchard (1971), Bjorndal (1982), Groombridge (1982), Carr et al. (1982), Bacon et al. (1984a), and Pritchard and Trebbau (1984) offer the most complete discussions of the present distribution of leatherback nesting grounds. Based on summaries presented by these authors, nesting in the western Atlantic occurs in Brazil, the Guianas (French Guiana, Surinam, Guyana), Venezuela, Central America (notably Panama and Costa Rica), and Mexico. In the West Indies, Trinidad and the Dominican Republic are relatively important (supporting perhaps 200-300 leatherbacks per annum), while nesting elsewhere in the eastern Caribbean is predictable but occurs nowhere in large numbers. There is considerable anecdotal information to suggest that nesting has declined at many Caribbean rookeries over the last several decades. However, with few exceptions (see Cambers and Lima 1990, Eckert 1992) documentation is lacking. Information concerning the historical status of western Atlantic populations is currently being compiled by the Wider Caribbean Sea Turtle Recovery Team and Conservation Network (WIDECAST), as part of an effort to draft sea turtle recovery action plans for each of 39 Caribbean governments.

In the eastern Atlantic, direct and/or indirect evidence of nesting is available from Angola, Sénégal, Liberia, Côte d'Ivoire, Ghana, Gabon, Togo, and Zaire. Fretey and Girardin (1988) report nesting between Pongara Point and Ndindi in Gabon, near the Congo border, and suggest a population size of 1276-2553 breeding females. In the Mediterranean, rare nestings are reported from Israel and the southern coast of Sicily. In the Indian Ocean, the

best known nesting grounds are in Sri Lanka and in Tongaland, South Africa. Small-scale nesting occurs in India, and larger aggregations are known in the Union territories of the Andaman and Nicobar Islands. In Indonesia, Sumatra and Java reportedly receive some nesting; Irian Jaya is still important. Thailand (especially Phuket) is important, Burma less so, and there is still some nesting in Malaysia. In the western Pacific, Papua New Guinea, the Solomon Islands, and Australia have nesting (see Eckert 1993 for review). The largest nesting colonies in the eastern Pacific are located in Mexico and Costa Rica. Nesting occurs on a smaller scale along the Central American coastline, Colombia, Ecuador and, possibly, Peru.

In the U. S., St. Croix (USVI) and Culebra (Puerto Rico) support the largest nesting colonies of leatherback turtles. Between 1981-1993, the total number of nests laid per year at Sandy Point NWR ranged from 82 (1986) to 345 (1992), or 18 to 55 females (McDonald et al. 1993). Sandy Point NWR receives 50-70 % of all leatherback nests in the USVI. Between 1983-1988, 11-52 nests (= 3-11 females) were laid each year at Manchenil Beach, St. Croix (Adams 1988). Shoy's Beach on the north shore of St. Croix receives \pm 20 nests per year and another 15 beaches receive a lower level of nesting (Eckert 1992). Between 1982-1988, 41 leatherback activities (both successful and unsuccessful nests) were reported from Buck Island off the northeast coast of St. Croix. One and nine crawls were recorded on West Beach, Buck Island, in 1989 and 1990, respectively; there was no leatherback nesting in 1991 (Zandy Hillis, USNPS, in litt., 11 May 1992). A half dozen beaches on St. John and St. Thomas rarely exceed one nesting leatherback each per year (Boulon 1987).

On Culebra Island between 1984-1990, 88-184 nests (= 12-27 females) were laid per annum (Tallevast et al. 1990). Resaca and Brava beaches receive 91-100% of the nesting on Culebra (Tucker 1988); nesting occurs elsewhere in Puerto Rico, but nowhere in large

numbers (Cintron and Cintron 1987). In 1991, 18 nests were reported at Playa Añasco, three nests at Mayagüez, and one nest at Cabo Rojo; between 1986-1991, 149 nests were reported at Luquillo (Benito Pinto, PR DNR, pers. comm.). Between 1988-1991, 9-26 nests per year were laid at Playa Humacao (Manuel Corbet, PR DNR, unpubl. data). An April-June 1991 survey of Vieques Island revealed 30 leatherback nests (B. Pinto, pers. comm.). Kontos (1986) documented 11 nests on Mona Island between 17 April-19 June 1985; there were no nests in 1986, five crawls in 1987, and none in 1988 or 1989 (Richardson 1990). Nesting on the Atlantic coast of Florida may sometimes approach that reported in the U. S. Caribbean, but nest density is considerably lower. The number of nests reported per year in Florida has risen with increasing survey effort from 18 nests in 1979 to 177 nests in 1992 (Meylan et al. 1995). Nesting north of Florida is very rare (see Schwartz 1977, Ruckdeschel et al. 1982). No nesting is reported from areas under U. S. jurisdiction in the Pacific Ocean (Eckert 1993).

Estimates of foraging population sizes are not available. Sightings and incidental catch data indicate that leatherbacks (adult or near adult size) are found in Alaska as far north as 60.34°N, 145.38°W and as far west as the Aleutian Islands (Hodge 1979, Stinson 1984). Documented encounters extend southward through the waters of British Columbia (Logier and Toner 1961, MacAskie and Forrester 1962), Washington and Oregon (Eisenberg and Frazier 1983, Brueggeman 1991), and California (Stinson 1984; Starbird et al. 1993). After analyzing some 363 records of sea turtles sighted along the Pacific coast of North America (from 29°45'N northward), Stinson (1984) concluded that the leatherback was the most common sea turtle in U. S. eastern Pacific waters. The species is also present in U. S. temperate Atlantic waters where recent efforts to calculate sea turtle population size using aerial survey techniques have been largely unsuccessful with regard to leatherbacks (Thompson 1988, Epperly et al. 1990), presumably because of a statistically inadequate

number of sightings. In the one published study that made an attempt to quantify density, Lohofener et al. (1988) reported 0.027 leatherbacks per 100 km² during periods of peak density offshore of Louisiana in October 1987.

A survey of the Cape Canaveral area conducted during March 1982-August 1984 reported that 94.5 % of all leatherback sightings (n = 128 total) occurred east of the 20 m isobath and 90.6 % occurred during the summer (Schroeder and Thompson 1987). An earlier (1979) aerial survey of the mid- and north-Atlantic areas of the U. S. Outer Continental Shelf between North Carolina and Nova Scotia showed leatherbacks to be present April to November throughout the study area (but most likely to be observed from the Gulf of Maine south to Long Island); peak estimates of relative abundances during the summer were in the hundreds (Shoop et al. 1981). The same study concluded that leatherbacks were observed more frequently in colder waters at higher latitudes during the summer than were other sea turtle species. In Cape Cod Bay, Massachusetts, sightings peak in August and September (Prescott 1988). Sightings peak progressively earlier in the year moving south along the eastern seaboard. In North Carolina public sightings of leatherbacks inshore and offshore are highest in May (Epperly et al. 1995). In South Carolina abundance peaks in mid-May, based on annual aerial offshore survey data reporting mixed groups (male, female; adults, subadults) moving north at this time of year (Sally Murphy, SC WMRD, pers. comm.).

It is not known at the present time whether leatherback populations under U. S. jurisdiction are stable, increasing, or declining, but there can be no doubt that some nesting populations (e.g., St. John and St. Thomas, USVI) have been virtually extirpated (Eckert 1992). Historical levels of nesting are not known at either Sandy Point NWR or Culebra Island, but data from continuing studies at these sites show annual fluctuations which do not appear abnormal and do not project a long-term decline. Even as these important nesting beaches

are protected, however, egg poaching remains a serious concern in both the USVI, and especially Puerto Rico. Furthermore, incidental catch and entanglement at sea are unquantified and potentially important sources of mortality.

Listing Factors

1. Present or Threatened Destruction, Modification or Curtailment of Habitat or Range

Nesting beaches around the world have been and are being degraded by residential and commercial coastal construction (particularly high density development), beach-front lighting, beach armoring (including stabilization structures such as seawalls, groins, and erosion barriers), mechanized beach cleaning, upland deforestation, sand mining, uncontrolled recreational use, vehicle traffic, litter, and the removal of stabilizing vegetation. General overviews of habitat alteration and its observed consequences are provided by Bjorndal (1982), Groombridge (1982), Coston-Clements and Hoss (1983), Bacon et al. (1984b), Hopkins and Richardson (1984), Raymond (1984), National Research Council (1990), and NMFS\FWS (1992). The potentially damaging effects on leatherbacks of chemical pollutants in the marine environment, such as industrial effluents, crude oil and pesticides, have yet to be determined.

At Sandy Point NWR, St. Croix, contemporary problems include egg poaching, domestic animals (dogs, horses), four-wheel drive vehicles, umbrellas and volleyball nets, cooking and bonfires, indiscriminate garbage disposal, rape, robbery, loud music, drinking, and drug use (Greg Hughes, USFWS, in litt., 12 May 1992). Litter such as discarded glass and metal has caused nesting attempts to be aborted at the Refuge (K. Eckert, pers. observ.) and vehicle traffic has resulted in death to 20 % - 25 % of hatchlings in nests which have been driven

over (Eckert et al. 1984). Many of these problems were eliminated when a policy of dusk to dawn closure was enacted in 1993, and later further reduced to weekend visitation only. Still, problems remain. For example, nesting females and particularly hatchling turtles are disoriented by coastal lighting and are often attracted, particularly on moonless nights, to the intense glare of elevated baseball field lighting in Frederiksted several kilometers to the north (K. Eckert, pers. observ.).

At the other major U. S. nesting ground, the development of Culebra's coastal zone poses direct threats to nesting and offshore habitats. Beach sand mining, nest compaction and hatchling death due to vehicular and pedestrian traffic, photopollution, sedimentation from upland clearing, and anchor damage have been documented at Brava, Resaca, and Zoni beaches (Teresa Tallevast, USFWS, in litt., 2 June 1992). On the main island of Puerto Rico, coastal zone development is a chronic threat, bringing increased traffic, litter, and artificial lighting. The recent construction of a pipeline at Playa Humacao, a beach regularly visited by leatherbacks, involved bright lights maintained on the beach while construction proceeded during the nesting season; not a single leatherback ventured ashore during this time (B. Pinto, pers. comm.). Similar problems plague leatherback nesting colonies throughout the Caribbean region; for example leatherback nesting beaches have been lost to sand mining in the BVI (Cambers and Lima 1990), St. Kitts and Nevis (Eckert and Honebrick 1992), St. Lucia (d' Auvergne and Eckert 1993) and Grenada (Eckert and Eckert 1990a), and elsewhere. Beach-front development is everywhere increasing, and brings with it a host of threats related to coastal construction and recreation.

Oceanic foraging grounds are also being degraded, primarily as a result of anthropogenic waste disposed at sea (e.g., UNEP 1984, CEE 1987). There is mounting evidence that the ingestion of plastic bags, presumably mistaken for jellyfish, is commonplace for leatherback

turtles and can be fatal. Eleven of 15 leatherbacks that washed ashore on Long Island, New York, during the summer of 1982 had plastic bags "totally blocking their stomach openings" (Anon. 1983). Ten of 33 dead leatherbacks washed ashore on Long Island between 1979 and 1988 had ingested plastic bags, plastic sheets, or monofilament (Sadove and Morreale 1989). Plastic bags and film were "common" in the intestinal tracts of leatherback carcasses discarded by fishermen in Peru (Fritts 1982). Mrosovsky (1981) reviewed stomach content data and found that in seven of 16 cases (44%), the leatherbacks examined had ingested plastic. Hughes (1974) reported that an adult female stranded in Natal, South Africa, "had its duodenal tract completely filled by a sheet of heavy plastic" measuring 3 m x 4 m. Near Rochelle, France, seven of eight leatherbacks examined had swallowed plastic (Duron and Duron 1980). These are not isolated data, and suggest that the ability of leatherback populations to recover depends not only on protection from direct harvest and the conservation of important nesting habitat, but also on the general health and cleanliness of the oceans.

Oil spills are a continuing source of concern and the extent of their effect on leatherbacks or their eggs has not been assessed. In September 1989, following Hurricane Hugo, a 42,000 gallon spill of #6 fuel oil (heavy crude oil) at the Water and Power Authority facility in Christiansted, St. Croix, left south coast beaches heavily oiled. Between March 1991 and March 1992, two more spills, both outside of U. S. waters, threatened U. S. nesting beaches. The first occurred on 6 March 1991, 13 nm north of Nevis, when the Trinidad-registered barge Vestabella, loaded with about 560,000 gallons of #6 fuel oil, sank in 600 m of water after a towing cable snapped; the initial oil slick was more than 30 miles long (Simmonds 1991). According to *The Daily News* (30 March 1991), a USVI newspaper, tar balls began appearing on St. John on 21 March and soon thereafter were reported from St. Thomas, St. Croix, Culebra, Vieques, and the main island of Puerto Rico. One year

later, on 15 March 1992, a pipe ruptured during ship-to-shore pumping of #6 fuel oil to a transfer station at St. Eustatius Terminal, Netherlands Antilles. One hundred barrels of crude oil were released to the sea in a slick that headed northwest out across the rich fishing grounds of the Saba Bank. Heavy seas broke up the slick before it entered U. S. waters, but tar balls eventually fouled the coast of Puerto Rico (Z. Hillis, pers. comm., 1992).

2. Over-utilization for Commercial, Recreational, Scientific, and Educational Purposes

There is virtually no international commerce in leatherback turtle products. Nonetheless, local commercial and subsistence exploitation is heavy in many parts of the world. The literature is replete with accounts of the harvest of eggs and/or adults in all major breeding areas. In the Western Atlantic, leatherbacks are widely taken for both meat and oil, the latter consumed for general vitality or employed medicinally for respiratory congestion. Tag returns indicate that this take occurs at least in part in populations that migrate between foraging grounds under U. S. jurisdiction and tropical nesting beaches outside U. S. jurisdiction. Prior to recent conservation programs involving indigenous peoples (Tambiah 1992), an estimated 80 % of nesting females were slaughtered during their attempts to nest on beaches in Guyana (Pritchard 1986). Ross and Ottenwalder (1983) reported that nearly 100 % of gravid females were harvested by local people for food in the Dominican Republic. The harvest of adults continues in the BVI (Cambers and Lima 1990), Grenada (Eckert and Eckert 1990a), St. Lucia (d'Auvergne 1992), Trinidad (Chu Cheong 1990), and elsewhere. In addition to harvest on tropical nesting beaches, there are several reports of leatherbacks shot or harpooned in temperate Atlantic (Brongersma 1972) and Pacific (Stinson 1984) latitudes.

The harvest of eggs also persists throughout the turtle's breeding range, including much of the Western North Atlantic. In Puerto Rico, where the collection of sea turtle eggs has been illegal for many years, Matos (1986) reported a "flourishing black market offering high prices for meat and eggs" and concluded that the illegal hunting of nesting females and eggs was a threat to population recovery. As recently as 1992 there was evidence of egg poaching at major U. S. rookery sites; i.e., Sandy Point NWR and Shoy's Beach on St. Croix (G. Hughes, pers. comm., 1992) and on Culebra (T. Tallevast, pers. comm., 1992). In Pacific Mexico, where leatherbacks feeding off the west coast of the U. S. presumably go to reproduce, egg collection is widespread despite laws prohibiting it (L. Sarti M., pers. comm., 1994). Further south, the harvest of eggs by local people for food or sale is so intensive that few nests laid from Costa Rica to Colombia can be expected to survive (Pritchard 1989). In Malaysia, a predominantly Islamic nation where the consumption of sea turtle meat is forbidden by religious custom, virtually 100% of the eggs were collected (Chan et al. 1985) prior to a 1989 moratorium on egg harvest. This exploitation is implicated in the steady decline of that population. Between 1967-1976 and 1984-1988, based on mean values, there was an 86% drop in the number of turtles visiting the well-studied Terengganu rookery (Chua 1988).

The effects of utilizing leatherback turtles for scientific purposes are not likely to negatively affect remaining populations. Leatherback research consists primarily of population surveys, hatchery programs, and other activities which do not involve the death or debilitation of individuals. In the United States, the use or take of leatherbacks for scientific purposes (e.g., captive diet studies, blood sampling, deploying remote equipment) is controlled by a permit system designed to protect endangered and threatened species. Live leatherbacks beyond the hatchling stage are difficult to maintain in captivity and are not intentionally captured for recreational or educational purposes.

3. Disease or Predation

The extent to which disease or parasites influence the survival prospects of wild leatherback turtles is unknown, but these are not considered serious threats. Wolke (1981) reported a case of enteritis. Ogden et al. (1981) diagnosed hematogenous septic arthritis and osteomyelitis involving the elbow, distal humerus, and proximal radius and ulna in a 135 cm adult stranded on the New England coast after being struck in the head by a boat propeller. Rothschild (1987) documented avascular necrosis of bone in Oligocene and Eocene members of the Dermochelyidae; the extent to which the phenomenon debilitates modern dermochelids is unknown. Analysis of adipose tissue, as well as samples of liver and pectoral muscle, from an adult male leatherback stranded in Wales in 1988 indicated that heavy metal values were not elevated above 'normal' background levels seen in other organisms (Davenport et al. 1990, Davenport and Wrench 1990).

Internal parasites include intestinal nematodes (species unidentified) and trematodes (*Pyelosomum* [= *Astrorchis*] *renicapite*, *Calycodes anthos*), amoebae (cf. *Entamoeba histolytica*), and gall bladder flukes (*Cymatocarpus* sp.) (Dunlap 1955, Yerger 1965, Brongersma 1972, Threlfall 1979). An external parasitic isopod (*Excorallana antillensis*) was reported on gravid leatherback nesting on St. Croix by Eckert and Eckert (1988). Several barnacle species appear to colonize leatherback turtles, including *Stomatolepas dermochelys* [= *elegans*], *Platylepas hexastylus*, *Balanus trigonus*, *Conchoderma auritum*, *C. virgatum*, and *Lepas anatifera* (see Eckert and Eckert 1988 for review).

Ghost crabs (*Ocypode quadratus*), vultures (*Coragyps atratus*), monitor lizards (*Varanus* sp.), genet cats (*Genetta* sp.), water mongooses (*Ichonyx* sp.), night herons (*Nyctanassa violacea*), gray foxes (*Urocyon* sp.), raccoons (*Procyon lotor*, *P. cancrivorus*), coatis (*Nasua*

nasua), fly larvae (*Megaselia scalaris*), locust larvae (family Acrididae), ants, and a variety of bacteria and fungi prey on eggs and hatchlings in the nest (e.g., Pritchard 1971, Fretey 1981, Whitmore and Dutton 1985). Exotic species, such as the wild pig (*Sus scrofa sulawensis*) in New Guinea (Bhaskar 1985) and the mongoose (*Herpestes auropunctatus*) in the Caribbean (Eckert et al. 1984), also consume eggs and hatchlings. Rates of depredation do not appear to pose an important threat to U. S.-nesting populations. Losses to non-human predators at Sandy Point NWR are estimated to be < 0.5% of annual productivity (Eckert et al. 1984). In contrast, at Les Hattes, French Guiana, the loss to domestic dogs of nearly 1000 hatchlings per night has been documented (Fretey 1981).

Crocodiles periodically consume nesting adults in Papua New Guinea (Lockhart 1989). Jaguars and tigers may have been historical predators on nesting adults mainly in the Guianas, but are less important today (Pritchard 1971, Henri Reichart, STINASU, pers. comm., 1994). At sea leatherback hatchlings are consumed by fishes, having been found in the stomachs of carnivorous fishes in the waters of French Guiana (Fretey 1981). Juveniles and adults are preyed upon by killer whales (*Orcinus orca*) (Caldwell and Caldwell 1969, Sarti et al. 1994), crocodiles (Pritchard 1981), and, presumably, sharks. Rates of depredation on any life stage in the open ocean are unknown.

4. Inadequacy of Existing Regulatory Mechanisms

In the United States, both national and international laws protect leatherback turtles from harassment, harvest, and commerce. According to the Code of Federal Regulations (50 CFR 17.11), the leatherback sea turtle is listed as Endangered throughout its entire range under the U. S. Endangered Species Act (ESA) of 1973, as amended (35 FR 8495; June 2, 1970). Similarly, the species is classified as Endangered in the International Union for Conservation

of Nature and Natural Resources' (IUCN) *Red Data Book*, where taxa so classified are considered to be "in danger of extinction and whose survival is unlikely if the causal factors continue operating" (Groombridge 1982). Leatherbacks are included on Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). The U. S. ratified CITES in 1974.

All federal agencies must ensure that actions authorized, funded, or carried out by them do not result in the destruction or adverse modification of habitat designated as Critical Habitat for marine turtles pursuant to Section 7 of the ESA. Critical Habitat has been designated by both the U. S. Fish and Wildlife Service (USFWS) and the National Marine Fisheries Service (NMFS). The beach at Sandy Point, St. Croix, became the first nesting beach for any marine turtle to be proposed as Critical Habitat (Federal Register, 23 March 1978; 43 FR 12050-12051) (Dodd 1978). In September 1978, FWS determined the perimeter of the Sandy Point peninsula (3.06 km in length and 0.32 km wide from MHT inland) to be Critical Habitat for leatherback turtles. In March 1979, NMFS designated the surrounding waters, following the 100 fathom contour from 17°42'12"N around the peninsula to 64°50'00"W, as Critical Habitat (see USFWS 1981). In September 1984, FWS purchased 2.4 km of this nesting beach, establishing Sandy Point NWR. With three species of sea turtle nesting at this site, Sandy Point is one of the most unique endangered species Refuges in the U.S.. Yet a manager was not hired until 1990, and the Refuge still lacks an informational kiosk and other interpretive materials.

The most important leatherback nesting beaches in Puerto Rico, namely Resaca and Brava on Culebra Island, were designated Critical Habitat in 1982 (effective 26 July), but the designation was for the protection of hawksbill, not leatherback, turtles (Dodd 1978). There is a National Wildlife Refuge on the island which includes a portion of the upland areas

behind the major leatherback nesting beaches (Playa Resaca, Playa Brava), but Culebra NWR does not encompass the beaches themselves. These two beaches are managed by the Commonwealth of Puerto Rico, with technical assistance and enforcement back-up provided by the USFWS (U. S. Department of the Interior). A paucity of proper land use planning and zoning, as well as a lack of rigorous or consistent enforcement of existing zoning and natural resource regulations by PR DNR and a corresponding inability or unwillingness on the part of the USFWS to enforce environmental policies, has had calamitous results at some nesting beaches, including uncontrolled coastal development, sand mining, erosion, and egg poaching.

Broad legal protection has been effective in eliminating egg and turtle poaching in the continental USA, but not so in Puerto Rico or the USVI. On the island of Puerto Rico, the theft of eggs for local consumption is nearly ubiquitous (Matos 1986, Cintron and Cintron 1987). Inadequate law enforcement is a widely recognized problem. Some progress has been made in Culebra. Egg poaching on this satellite island was once described as "extensive and unrelenting" (Carr 1978), but has noticeably declined on some beaches as a result of nightly patrol and nest protection programs initiated in 1984 (Tucker 1988). Egg theft also continues in the USVI, principally on St. Croix. The Virgin Islands Department of Planning and Natural Resources' Law Enforcement Division lacks sufficient funds and manpower to fully cover the beaches and phones are unmanned on weekends when most people observe violations (Z. Hillis, *in litt.*, 11 May 1992). An ongoing egg protection program at Sandy Point NWR has greatly reduced (but not eliminated) egg poaching and has resulted in substantial increases in annual productivity (Eckert and Eckert 1990b). Regulations for the protection of leatherbacks and their eggs are adequate, but vigilant enforcement and more visible public awareness campaigns are clearly needed.

At sea, leatherbacks are vulnerable to incidental capture. In addition to the take of leatherbacks by the shrimp trawling industry (e.g., Henwood and Stuntz 1987), NMFS has information that leatherbacks are incidentally taken in other types of commercial fishing gear in U. S. waters, particularly pelagic longlines and drift/gillnets (Wetherall et al. 1993). The agency has issued a Biological Opinion on this subject, evaluating a variety of fisheries with regard to the incidental catch problem (NMFS 1989). The use of turtle excluder devices (TEDs) is mandatory in the U.S. (Crouse 1993) and federal law also requires that foreign countries exporting shrimp to the U.S. enact and enforce sea turtle conservation measures, including the use of TEDs (Ple 1990). Not all TEDs will release leatherbacks because of their large size. There is, however, a provision in the U. S.-prepared regulations to close areas to trawling or greatly restrict tow times to protect turtles such as leatherbacks (C. Oravetz, pers. comm., 1992).

Recent progress has been made on the issue of incidental capture in waters beyond national jurisdiction. On 20 December 1991, the United Nations General Assembly passed a driftnet resolution co-sponsored by the United States and Japan that called upon all nations to reduce fishing effort in existing large-scale pelagic driftnet fisheries by 50 % by 30 June 1992 and to fully implement a global moratorium on all such driftnets by 31 December 1992. The resolution passed by consensus. In addition to incidental capture, leatherbacks are also threatened by marine pollution. A variety of legal regimes exist to address the problem of ocean dumping both regionally and internationally (see Bean 1987, Lentz 1987). These include the Law of the Sea Convention, The London Dumping Convention, the International Convention for the Prevention of Pollution from Ships (MARPOL), and a host of regional agreements, such as those sponsored by the UNEP Regional Seas Programme. Despite achievements in these areas, the problems persist on a grand scale. International support for

stringent regulations governing indiscriminate fishing and ocean dumping is needed, as is a mechanism for effective enforcement.

5. Other Natural or Manmade Factors

Incidental capture and entanglement at sea in active and abandoned fishing gear present serious problems for leatherback turtles worldwide (e.g., Brongersma 1972, Balazs 1985, Wetherall et al. 1993). Strandings reported from U. S. Atlantic, Gulf of Mexico, and Caribbean coasts increased 123 % in 1987 (n = 134) over 1986 (n = 60) (Schroeder 1988). Strandings declined in 1988 (n= 79), but have since risen again (n= 104 in 1989, n= 102 in 1990, n= 118 in 1991, n= 123 in 1992) (Wendy Teas, NMFS, pers. comm.). In April 1991, 30 leatherbacks washed ashore dead in Georgia, nearly equally the total of 39 during the previous 11 years; shrimp trawling was implicated (Shoop 1991). As of 1 June 1992, a record number of 28 adult and subadult leatherbacks had stranded on the beaches of South Carolina; gillnets and trawlers were the most likely causal agents. Six of the South Carolina turtles were fresh enough to necropsy. None of these had obvious injuries and all had a gut full of *Stomolophus* jellyfish. In three cases, froth in the trachea suggested recent drowning. In addition, some stranding network reports noted post-mortem and/or ante-mortem injuries, including skull fractures (Sally Murphy, SC WMRD, pers. comm.).

Summarizing a decade (1977-1987) of data, Prescott (1988) implicated entanglement (mainly in lobster pot lines) in 51 of 57 (89 %) adult leatherback strandings in Cape Cod Bay, Massachusetts. Goff and Lien (1988) reported that of 20 leatherbacks encountered off the coasts of Newfoundland and Labrador (1976-1985), 14 (70 %) were entangled in fishing gear, including salmon nets, herring nets, gill nets, trawl lines, and crab pot lines. Gill net entanglement is also suspected in the waters of New England, based on observations of

leatherbacks with cuts, severed limbs, or chafing marks (National Research Council 1990). Further south the species is a rare but regular casualty of shrimp trawls, and some individuals have become "entangled in the gangion or caught on the hook" of longlines set in the northern Gulf of Mexico (Hildebrand 1987), southeastern U. S. (Witzell 1984), and northern Caribbean (Cambers and Lima 1990, Tobias 1991). On 21-22 March 1992, five 300-foot gill nets were legally set as close as 10 m from shore in Critical Habitat off Sandy Point NWR. There is no evidence to indicate that gravid females were caught during this incident, but the fishermen involved readily admitted that leatherbacks had been ensnared in the past and killed in order to save the net (G. Hughes, pers. comm., 1992).

In the eastern North Pacific leatherbacks become entangled in gill nets off the coasts of California, Oregon, and Washington (Stick and Hreha 1989, Scott Eckert, HSWRI, pers. comm., 1994). Eleven leatherbacks were captured in gill nets by a single fishermen from Bahia de la Paz, Baja California, between 1985 and May 1987 (Alvarado and Figueroa 1990). Both adults and juveniles are occasionally caught in tuna purse-seines operating in the eastern tropical Pacific (S. Eckert, pers. comm., 1994). A very young individual (about 15 cm carapace length) was captured in a purse-seine in April 1976 about 180 nm west of San José, Guatemala (Robert Pitman, NMFS, pers. comm.). Balazs (1982) documented the death of leatherbacks in pelagic drift nets. Further south, Frazier and Brito Montero (1990) reported that "at least several hundreds" are captured per annum incidental to the swordfish industry in Chile. Finally, leatherbacks are struck, sometimes fatally, by propellers (Brongersma 1972, Stinson 1984). Leatherbacks are also incidentally captured in the water intakes of industrial facilities, such as power plants. Three were trapped in the St. Lucie Power Plant (Hutchinson Island, Florida) in 1978 and two more in 1981 (Roithmayr and Henwood 1982); a total of eight were reported between 1976 and 1988 (Ernest et al. 1989). Subsequent captures occurred in 1989 and 1992, one turtle in each case. The size range of

leatherbacks involved has been 112.5-150.0 cm SCL; all were released unharmed (Eric Martin, Applied Biology, pers. comm., 1992).

On the nesting beach, storm events and seasonal erosion degrade or destroy nesting beaches and can result in egg losses ranging from < 2.5% in Malaysia to 50% or more in the Guianas and the U. S. Virgin Islands (Mrosovsky 1983, Eckert 1987). Eggs may also be destroyed by females digging into nests constructed earlier in the season (Fretey 1986). Shimmering inland lagoons, or debris that blocks the ocean's bright horizon, can disorient hatchlings and preclude their passage to the sea (Mrosovsky 1983). Adult females in French Guiana are sometimes unable to reach the sea after nesting, having become trapped behind accumulations of dead trees, impaled against a root or branch, or held prisoner between stumps deeply embedded in the sand (Pritchard 1971, Fretey 1977, 1981). Females disoriented by an inland lagoon or swamp may mistake its reflection for that of the sea and orient toward it after nesting. Turtles mired in the soft mud of a mangrove swamp or river delta can die from exposure, exertion or asphyxiation.

Conclusion

While the leatherback sea turtle enjoys complete protection in the U. S., the species is often inadequately protected elsewhere. The direct and/or indirect harvest of adults continues in all major and most minor breeding grounds. This despite the fact that the IUCN considers the species Endangered throughout its global range; that is, "in danger of extinction and whose survival is unlikely if the causal factors continue operating" (Groombridge 1982).

In Mexico, the largest breeding colony of leatherback turtles in the western hemisphere has suffered enormous losses of both adults and eggs in recent decades (Pritchard 1982, Eckert

1993). The extent to which the recent ban on sea turtle harvest in Mexico (Aridjis 1990) will ultimately mitigate this situation remains unknown. Referring to Irian Jaya, host to the largest breeding colony of leatherback turtles in the eastern hemisphere, Bhaskar (1985) wrote, "The proliferation of outboard engines in the area in the last decade has made the once remote nesting beaches easily accessible to egg collectors who remove clutches by the boatload. On less favored beaches ... to the southeast, nesting leatherbacks are killed for food by inhabitants newly settled on the coast. In years to come, the turtle population will undoubtedly be under even greater human pressure as transmigration of Indonesian families into Irian Jaya continues." This prediction appears to have come to pass. Recent reports indicate an alarming decline in the nesting population at the Jamursba-Medi site (Betz and Welch 1992).

Pritchard (1981) reviewed the status of leatherback turtles in the Solomon Islands and concluded that there had been a gradual decline over the years, ominous but difficult to document. While not all leatherback populations are declining, Pritchard's words accurately describe the situation throughout much of the species' range. Perhaps it is fortunate in a backhanded way that in some areas these seemingly innocuous gradual declines have matured into population collapse. By examining these disasters, by evaluating cause and effect, we may be able to avert similar catastrophes elsewhere. One case in point is Terengganu, Malaysia, where nesters arrived by the low thousands per annum 25 years ago and now arrive in the tens. The collapse was precipitated not by the harvest of turtles, but of eggs. As a generation of breeders has died out, there has been no significant recruitment. Another illustrative example is the British Virgin Islands, where a subsistence harvest of gravid females for meat and oil has extirpated a once thriving population. The nature of sea turtle life history allows nesting turtles and/or eggs to be over-harvested for decades from large populations, seemingly without consequence (Mortimer 1995). By the time declining trends

are statistically perceived amid the background noise of normal annual fluctuations, a population may be committed to a downward spiral from which it is not capable of recovering. Malaysia recently implemented a moratorium on the harvest of leatherbacks and their eggs and the BVI is attempting to do the same. In both cases it is likely that the decision has come too late. Many leatherback populations have been seriously over-harvested in past decades and will need wise stewardship into the next century if they are to avoid potentially irreversible declines.

Causal factors operative in the marine environment have not been satisfactorily quantified but appear serious. Thompson et al. (1990) concluded that intensifying human utilization of deeper offshore waters is placing all life history stages of the leatherback turtle at "increasing risk". All indications are that debilitation, particularly of large juveniles and breeding age adults, due to entanglement in and ingestion of persistent marine debris is high. In addition, this status review has summarized incidental capture and drowning in a wide variety of fishing gear. On the basis of these data it would not be unreasonable to conclude that the status of foraging populations occurring in U. S. jurisdiction has worsened since protection was first afforded the species under the Endangered Species Act nearly 20 years ago. It is noteworthy that animals foraging in the U. S. include juveniles which will eventually recruit into the breeding aggregations of the western Atlantic and eastern Pacific, where they will (should present circumstances prevail) face considerable threat at the nesting grounds. The best available commercial and scientific data indicate that the leatherback sea turtle (*Dermochelys coriacea*) should remain listed as Endangered throughout its range pursuant to Section 4 of the Endangered Species Act of 1973, as amended.

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Hawksbill Sea Turtle, *Eretmochelys imbricata*

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Biological Background

Hawksbill sea turtles (*Eretmochelys imbricata*) are distinguished from other sea turtles by two pairs of prefrontal scales; thick, posteriorly overlapping carapace scutes; four pairs of costal scutes, the anterior-most not in contact with the nuchal scute; and two claws on each flipper. Some scute variation occurs in both adults and hatchlings (Limpus et al. 1983). The carapace is typically serrated along the posterior margin (becoming less so with age) and is "tortoiseshell" in color and pattern, the dorsal laminae showing radiating streaks of brown, black and amber. Carapace color is geographically variable and may also change with age (see Witzell 1983 for review). The head is relatively narrow, the beak tapers to a point, and the maxilla projects slightly beyond the mandible. The scales of the head are dark brown with pale yellow margins. Both adults and hatchlings have a normal tetrapod gait while on land, with alternating movements of opposing flippers (Pritchard 1979). Two subspecies, *E. i. imbricata* in the Atlantic Ocean and *E. imbricata squamata* in the Pacific Ocean, have been described on the basis of differences in coloration and carapace shape. The criteria have proven unreliable in distinguishing the two forms, however, and subspecific designations are rarely used (Meylan 1984a). A general synopsis of the biology of the species is provided by Witzell (1983) and more recent, in-depth discussions can be found in Pritchard and Trebbau (1984) and Groombridge and Luxmoore (1989).

Mean straightline carapace length (SCL) of adult females ranges from about 66 to 86 cm worldwide (Witzell 1983). Meylan (1984b) reported that adult females in the Caribbean measured 62.5-91.4 cm SCL. Weight is typically to 80 kg (Pritchard et al. 1983), with an historical record of a 280 lb [127 kg] individual caught at Great Sound, Grand Cayman (Carr 1952). Nietschmann (1981) observed little dimorphism between adult males and females in Caribbean Nicaraguan waters. He reported average carapace lengths of 77.8 cm (range 71.4-85.1, n= 17) and 76.5 cm (range 62.5-87.0, n= 32), respectively, and average weights of 53.4 kg and 54.2 kg, respectively. Adult males are distinguished by a long, thick tail that

extends well beyond the carapace margin and well developed, recurved claws on the fore flippers (Witzell 1983). In the U.S. Caribbean, the curved carapace length (CCL) of nesting females averages about 87 cm (n= 61) (Hillis and Mackay 1989, Richardson 1990). At Buck Island, situated off the north shore of St. Croix, the CCL of nesting females ranged from 78.7-100.0 cm during 1988-1991 (Zandy Hillis, USNPS, *in litt.*, 11 May 1992). Hatchlings emerging at Buck Island average 41 mm SCL and weigh 14.0-17.5 g (Hillis and Mackay 1989). At Sandy Point National Wildlife Refuge (NWR), St. Croix, egg diameter averages 39.9 mm (n= 36 eggs); hatchlings average 42.4 mm SCL and weigh 18.5-19.5 g (n= 60 eggs) (Eckert, unpubl. data). Morphometric data are not available for this species in the U.S. Pacific.

Hawksbills are distributed circumglobally and appear to be the most confined of all the sea turtles to tropical waters, although individuals are sometimes encountered in northern latitudes (e.g., Ireland: Penhallurick 1990). The species is often characterized as sedentary, or as having "given up migration" (Hendrickson 1980). This stereotype has persisted in part because of early reports (Carr and Main 1973) that carapace color and morphology varied by island (or island group) in the southwestern Pacific and were unique, allowing indigenous peoples to discern distinct populations over rather restricted geographic areas. There is skepticism, however, that such uniformity exists (Limpus et al. 1983). The hypothesis that the species is sedentary is further eroded by records of long distance movements of several hundred to several thousand km (e.g., Carr and Stancyk 1975, Nietschmann 1981, Vaughan 1981, de Silva 1982, Meylan 1982, Parmenter 1983, Marcovaldi 1991, Hillis 1994) as well as data obtained via satellite telemetry revealing long distance post-nesting movements (Groshens and Vaughan 1994). In the Caribbean region, juvenile hawksbills tagged in the USVI have been recovered in the BVI, Puerto Rico, St. Lucia, and St. Martin (Boulon 1989). Hawksbills tagged in Antigua later have been recaptured in Dominica (Fuller et al. 1992) and St. Kitts (K. Orchard, SCHS, pers. comm., 1994). On 20 July 1990 a juvenile hawksbill (74 cm SCL) tagged six months before at the Biological Reserve of Atol das Rocas in Brazil, was captured and killed in Dakar, Senegal, some 2,300 miles away (Marcovaldi 1991). It is likely that additional tagging in nesting and foraging habitats will corroborate the hypothesis that migratory behavior occurs in hawksbills.

Hawksbills forage in coral reefs and other hard-bottom habitats in open bays and coastal zones throughout the tropics and, to a lesser extent, the subtropics, including Florida, the

Gulf of Mexico, the USVI, Puerto Rico, Hawaii, and the U.S. Pacific territories. Despite a wide variety of foods consumed (see Witzell 1983), recent studies indicate that hawksbills specialize on sponges in the Caribbean, and predominately on two orders of Demospongia (Meylan 1988, vanDam and Diez 1994, Vicente 1994). Ten species of sponges accounted for 79.1 % of the dry mass of all sponges identified in the stomachs of hawksbills from seven Caribbean countries (Meylan 1988). The predominance of specific taxa in the digesta suggests a degree of selectivity, perhaps related to distinctive properties of the sponges with respect to spongin and collagen (Meylan 1985). Few data are available from the Pacific, but Balazs (1978a) reported that the stomach and intestines of a dead hawksbill (75.6 cm SCL) entangled in a monofilament gillnet in Kanehoe Bay, Hawaii, were filled with three kinds of unidentified sponges. Hawksbills are solitary, and intraspecific competition for food resources has not been documented. Illness and even death have been attributed to the consumption of hawksbill meat in Sri Lanka, China (Taiwan), Philippines, Indonesia, Papua New Guinea, and Australia (Torres Strait), as well as Central Pacific and Caribbean islands (Halstead 1970, Hashimoto 1979, Márquez 1990). Most authors speculate that the toxins originate in food items consumed by the turtle.

Mating occurs on the surface in shallow waters adjacent to the nesting beach and may last several hours; polyandrous breeding behavior is implied (Witzell 1983). Gravid females most often nest on isolated (typically insular) nesting beaches with well developed supralittoral vegetation. Nests are commonly, but not universally, placed amongst woody vegetation (Mortimer 1982, Ryder et al. 1989). During 1984 on Mona Island, Puerto Rico, all nesting attempts except one occurred 1-6 m into the vegetation, generally *Suriana maritima* (Olson 1985). Nesting is mainly nocturnal, but some populations nest infrequently (Limpus et al. 1983) or nearly entirely (Diamond 1976) in the daytime. The nesting sequence consists of landing, selecting a nest site, clearing the site and excavating a shallow body pit, excavating the nest chamber, egg laying, refilling the nest chamber and concealing the site, and returning to the sea. The entire procedure requires an average of 130 min at Buck Island, USVI (Hillis and Mackay 1989) and 126 min (range 72-196, n= 20) in Antigua (Jim Richardson, Univ. Georgia, pers. comm.), the lengthiest stages being nest excavation and site camouflage. Average clutch size (yolked eggs) varies geographically and appears to be strongly correlated with mean carapace size among rookeries (Limpus et al. 1983, Witzell 1985). Non-yolked eggs are uncommon, but in some cases have comprised up to 39 % of the clutch (Hirth 1980).

Nesting is seasonal, but the season is often extended and at a few localities nesting may occur throughout the year with one or two peaks (Groombridge 1982). Long-term nesting studies are rare, but one such study began on Cousin Island, Seychelles, in 1973, where about 25 females arrive to nest each year. Cousin Island hawksbills deposit an average of four clutches (mean= 182 eggs) per season (September-March) at intervals of 14-18 days and eggs incubate for 47-77 days (mean= 60 days, n= 16) (Diamond 1976). Brooke and Garnett (1983) later reported a mean clutch size of 161 eggs (the variation apparently due to different approaches to counting egg shells), an annual clutch frequency of 2.76 (as opposed to four), with remigrants more productive and neophytes slightly less so, and an average remigration interval of 2.67 yr. In the western hemisphere, 246 nesters were tagged at Tortuguero, Costa Rica, between 1955 and 1983. Data from this long-term study suggest that clutches (mean= 158 eggs) are deposited at average intervals of 16.4 days (n= 28) and incubate for 56-61 days (mean= 58.5 days, n= 6) before hatching. Because of deficiencies in beach coverage, clutch frequency and remigration intervals were not calculated (Bjorndal et al. 1985), but an earlier report noted that the most commonly observed remigration interval at Tortuguero was three years (Carr and Stancyk 1975).

An ongoing study at Jumby Bay, Antigua, unique in having 100 % of the females tagged, provides the most complete record of hawksbill reproductive ecology available. Some nesting occurs in every month of the year, but the major season is mid-June to mid-November. Females arrive at the nesting beach asynchronously. In 1988, 39 individuals deposited an average of 4.4 clutches (mean= 147 eggs/clutch, range 70-203) each at mean intervals of 14.8 days during June-November; the modal number of nests per turtle was five. Mean incubation time was 68 days and hatch success averaged 85 % (Corliss et al. 1989). Similar results were obtained in 1987 and 1989, when clutch frequency averaged 4.8 and 4.7 clutches per female and mean hatch success was 79 % and 84 %, respectively (J. Richardson, unpubl. data). Based on this systematic field effort, it is clear that stock assessment formulae relying on annual clutch frequencies of fewer than five per turtle are in danger of overestimating total numbers of nesting females (Richardson et al. 1989). Remigration to the Jumby Bay site is high, with 95% of the 1987 cohort (20/21) returning again in 1989 (n= 16) or 1990 (n= 4) and 66 % of the 1988 cohort (26/39) returning again in 1990 (n= 11) or 1991 (n= 15) (J. Richardson, unpubl. data).

Hawksbills are studied on beaches in the U. S. Caribbean at Virgin Islands National Park (St. John), Buck Island Reef National Monument (St. Croix), and Mona Island (Puerto Rico). The nesting season on St. John extends from June to December (peak: August-September) and clutch size and incubation average 141.6 eggs (n= 39) and 64 days (n= 28), respectively (Small 1982). Annual reports for the Buck Island project indicate that ca. 15-30 hawksbills nest each year, annual mean clutch size ranges from 137.3-153.4 eggs (n= 262 nests), inter-nesting intervals average 14-15 days (n= 77), and peak nesting occurs in July-September. Annual hatch success for nests that reach full-term without obvious disturbance from predators, erosion, or poachers ranges from 77 % to 83 %. On Mona Island there have been 738 nesting records reported (1113 estimated) during seven survey seasons since 1974, equivalent to some 36 nesting females per annum (Richardson 1990). The Mona Island data show that clutch size averages 141.0 (1989) to 157.6 (1984) eggs, inter-nesting intervals are typically 14-15 days, incubation requires 47 to 63 days, and roughly 75 % of the nests are laid during August-November (Richardson 1990). Similar studies have not been undertaken in the U.S. Pacific.

Mrosovsky et al. (1995) evaluated the effect of incubation temperature on sex determination in hawksbill hatchlings. Incubation temperatures warmer than approximately 29.2°C produced females, while cooler temperatures produced males (Mrosovsky et al. 1995). Dalrymple et al. (1985) reported a heavily male-biased sex ratio in a Florida nest that was laid late in the nesting season (25 October 1981) and incubated for some 91 days; maximum ambient temperature rarely reached 30°C. Once hatchlings leave the natal beach, virtually nothing is known of their distribution, abundance or survival. A brief study conducted in Puerto Rico suggests that hatchlings orient toward open ocean once they enter the surf (Hall 1987). There is some evidence that hatchlings and post-hatchlings may associate with *Sargassum* weed in the pelagic zone (Meylan and Carr 1982, Carr 1987). Juveniles of various size classes can be observed in hard bottom habitats throughout the tropics (in the U.S. they are predictably found as far north as southeastern Florida and the northern Gulf of Mexico), but no systematic study of their distribution, behavior, or survivorship has been undertaken.

Data from the capture and recapture of hawksbills in St. Thomas, USVI indicate that wild juveniles (size range 27.4-60.7 cm SCL) grow an average of 3.36 cm per year (Boulon 1983), while juvenile green turtles (size range 26-62 cm SCL, n= 41) in the same habitat

average 4.8 cm per year (Boulon and Frazer 1990). In contrast, juvenile hawksbills (40-70 cm SCL, n= 4) in the southern Bahamas grew at a rate of 2.4-5.9 cm per year, slightly faster than co-occurring green turtles (n= 62) in comparable size classes, and increased in mass at a rate about 1.5 times greater than did the green turtles (Bjorndal and Bolten 1988). Mature females grow at an average rate of 0.3 cm per year at Tortuguero, Costa Rica (Bjorndal et al. 1985). In the Pacific, nearly two decades (1969-1988) of systematic study in the Great Barrier Reef (GBR) of Australia have shown that (a) individuals recruit to the GBR feeding grounds at a minimum of 35 cm CCL, (b) there are significant differences in growth rate among size classes, with the maximum mean growth rate (2.17 cm/ year) recorded for turtles in the 50-60 cm CCL range, and (c) a small turtle (35 cm CCL) recruiting to the feeding grounds can be expected to begin breeding about 31 years later (Colin Limpus, Queensland Turtle Research, pers. comm., 1992). Age at maturity is unknown.

Population Size

Nesting beaches are distributed circumglobally, roughly from 30°N to 30°S, and can be identified in the Eastern and Western Atlantic, Eastern (rare), Central and Western Pacific, and Indian Ocean; the non-nesting range is equally extensive (Witzell 1983). There are no world population estimates for hawksbill turtles, but a minimum of 15,000 to 25,000 females are thought to nest annually in more than 60 geopolitical entities (Groombridge and Luxmoore 1989). Nesting usually occurs at low densities. Moderate population levels appear to persist around the Torres Strait islands, in the Red Sea ["possibly 500 nests yearly", Frazier and Salas 1984] and Gulf of Aden, and probably around the Arnavon Islands (Solomons), northern Australia, Palau, Persian Gulf islands, Oman, and parts of the Seychelles (Groombridge 1982). In a more recent review, Groombridge and Luxmoore (1989) list Papua New Guinea [but see Ehrenfeld 1990], Queensland, and Western Australia as likely to host 500-1000 nesting females per year, while Indonesia and the Seychelles may support > 1000. The largest known nesting colony in the world is located on Milman Island, Queensland, Australia where Loop (1995) tagged 365 hawksbills nesting within an 11 week period. With the exception of Mexico (and Cuba?), nearly all Wider Caribbean countries are estimated to receive < 100 nesting females per year (Meylan 1989).

Recent surveys have documented relatively large nesting colonies on the shores of the Yucatan Peninsula (see Frazier et al. 1993). As many as 800-1000 nests are laid annually

from Isla Holbox (Quintana Roo) to Isla Carmen (Campeche) (Richard Byles, USFWS, pers. comm.). A recent review by Frazier (1995) indicated 150 to > 300 nests per year in Campeche (Isla Aguada-Chenkán; possibly this is only 50 % of the annual total), > 50 nests at Celestun-El Palmar, 200-400 nests at Las Coloradas-El Cuyo, and about 200 on Isla Holbox. Regionally important colonies outside of Mexico exist in Nicaragua, Cuba, the San Blas Islands and Bocas del Toro region of Panama, Mona Island (Puerto Rico), and the Grenadines (Hopkins and Richardson 1984), as well as on the Manabique Peninsula, Guatemala (Rosales-Loessener 1987), near Manatee Bar, Belize (Smith 1992), and on Long Island, Antigua (Corliss et al. 1989). Groombridge and Luxmoore (1989) maintain that about half of the known nesting populations are known or suspected to be in decline; in particular, "the entire Western Atlantic-Caribbean region is greatly depleted." Prolonged over-exploitation for the international tortoiseshell trade and the widespread collection of eggs are implicated in the species' demise.

In the U. S., nesting occurs in the Atlantic (Florida), Caribbean (Virgin Islands, Puerto Rico) and Pacific (Hawaii, insular Pacific territories), but has been poorly quantified in most areas. Observed nesting in Florida is rare at 0-2 nests per annum (1979-1990; Anne Meylan, FL DNR, pers. comm.). Florida may support a slightly higher level of nesting, but nesting censuses are accomplished primarily by morning track counts, and hawksbill tracks may not always be differentiated from those of loggerhead turtles (*Caretta caretta*). In the USVI, Zullo (1986) reported 16-49 nests per annum in Virgin Islands National Park, St. John. In 1991, 64 crawls were reported from Cocolobo Beach and 11 more from elsewhere on St. John (Z. Hillis, in litt., 11 May 1992). As many as 50 nests per annum are reported for St. Thomas, but these are reported only opportunistically by fishermen and surely represent a gross underestimate (Ralf Boulon, USVI Div. Fish Wildl., pers. comm.). A summary of data available data for St. Croix suggests that nesting occurs on nearly 50 beaches, with 200 or more nests laid per year; Sandy Point NWR alone receives 10-20 nests per year and Isaac's, Jack's, and Coakley Bays may receive 10-30 nests each (Eckert 1992). Data from Buck Island (1988-1991) reveal 60-99 confirmed nests per year out of 158-240 crawls (Z. Hillis, in litt., 11 May 1992).

In Puerto Rico and its offshore cays, there are about 275 miles of sandy beaches (Cintron and Cintron 1987), making systematic surveys of hawksbill nesting difficult. In 1981, 22 nests were reported on Vieques Island (Pritchard and Stubbs 1982). A decade later, systematic

survey of this satellite island was initiated by the Puerto Rico Department of Natural Resources and the U. S. Department of the Navy; 25 hawksbill nests were recorded in 1991 (Benito Pinto, PR DNR, pers. comm.). Between June 1975 and June 1977, 73 nests were reported from "the Culebra group" (Carr 1978). In 1985, 23 nests were reported on Culebra Island (Tony Tucker, pers. comm. *in* Meylan 1989); one nest was reported in 1987 and 48 between 1989-1991 (B. Pinto, pers. comm.). Over 15 years an average of 160 nests per year were laid on Mona Island (1974, 1984-1989; Richardson 1990); 159 nests were laid in 1991 (B. Pinto, unpubl. data). Nesting on the mainland of Puerto Rico remains largely unquantified, but a few beaches (i.e., Piñones, Luquillo, Humacao) have been surveyed since 1985. Between 1988-1991, 16-27 hawksbill nests per year were laid at Playa Humacao (Manuel Corbet, PR DNR, unpubl. data). Only one nesting attempt has been reported from Luquillo (B. Pinto, pers. comm., 1992).

In Hawaii, nesting is known to occur on the main islands between July and November, including the sites of Halape and Apua Point at a remote location in the Hawaii Volcanoes National Park. The most consistently used sites seem to be at Kamehame Point (Hawaii) and on a black sand beach at the river mouth of Halawa Valley (Molokai); not more than a dozen females nest per year on all beaches combined (NMFS 1992). The Palau nesting population of hawksbills is the largest in Oceania north of the equator; nesting is concentrated on the small beaches of the Rock Islands between Koror and Peleliu islands (Maragos 1991). This population is severely stressed by chronic egg poaching and the hunting of turtles for jewelry and crafts (Maragos 1991). Palauan fishermen are unanimous in their opinion that both green and hawksbill turtles are "far less abundant" than they were 10-20 years ago (Johannes 1986). An estimated 100 females (hawksbills and green turtles combined) nest per year in American Samoa; many are harvested and populations have "seriously declined" throughout the territory (Tuato'o-Bartley et al. 1992). Rose Atoll was designated a National Wildlife Refuge in 1974 and an unquantified (probably low) level of hawksbill nesting occurs there (Balazs 1978b). Hawksbill nesting is reportedly of very little significance in Guam or the Northern Mariana Islands (Pritchard 1982; Groombridge and Luxmoore 1989). No nesting has been recorded in the unincorporated U.S. Pacific island territories (Eckert 1993).

Excluding the U.S. Pacific where firm data are virtually non-existent, the United States (Caribbean/Atlantic) probably supports a minimum of 650 nests per annum or, based on annual average clutch frequency of five nests per female (Richardson et al. 1989), perhaps

130 nesting females. It is useful to remember that estimates of population size ultimately depend not only on regular surveys of potential habitats, but on our ability to distinguish a successful nesting from an unsuccessful attempt. Crawl effort (total crawls/nests) varies temporally and geographically. In the Seychelles, 56.2 % of crawls were judged to have resulted in egg-laying (40.1 % were confirmed nests) in 1973-1974, whereas in 1974-1975, 62.0 % were so judged (57.2 % confirmed) (Diamond 1976). On Campbell Island, Australia, 77 % of crawls resulted in egg-laying (Limpus et al. 1983). On Mona Island, about half the crawls observed, on average, result in egg deposition (39 % - 76 %; Richardson 1990). At Pasture Bay, Antigua, 60% of 1987 activities were confirmed nests, 68 % in 1988 (Corliss et al. 1989). Between 1985 and 1990 on Buck Island, USVI, 42.2 % to 79.0 % of crawls were judged to be nests, while the proportion later confirmed to have eggs varied between 13.4 % and 55.6 % per year (Hillis and Mackay 1989, Z. Hillis, pers. comm., 1992).

Far fewer data are available regarding foraging population size and distribution. In waters adjacent to the continental United States, hawksbills have been reported in every state from Texas to Massachusetts, with the exceptions of Connecticut and Maine (Woodard 1980). While opportunistic foraging, especially during the summer months, is likely to occur along much of the eastern seaboard and northern Gulf of Mexico, the species is described as "predictable" only along the southeast coast of Florida from Palm Beach south into the Florida Keys (A. Meylan, pers. comm.). The Florida records are mainly of juveniles and subadults, although adults are occasionally reported, and the individuals involved are few in number. In the U. S. Caribbean, foraging is generally associated with insular shelf reefs. Preliminary observations from field surveys conducted in the USVI and Puerto Rico indicate that juveniles are typically found in water < 18 m deep, mostly associated with fringing reefs around the coastline, while adults forage in deeper waters on the shelf (R. Boulon, pers. comm.). Juveniles are less common than adults and may tend to forage over wider territories. Juvenile hawksbills tagged in the USVI have been subsequently recovered elsewhere in the Western Atlantic (Boulon 1989). Studies in progress on the ecology and population dynamics of immature hawksbills at their Mona Island, Puerto Rico foraging ground is focusing on the turtles' growth rates, diet, sex ratios, and genetic relationship in comparison with nearby nesting populations (vanDam and Diez 1994).

In the U.S. Pacific there are no hawksbill sightings off the west coast of the continent (Stinson 1984, Scott Eckert, Hubbs-SWRI, pers. comm., 1994). The Hawaiian population is small and only known to occur in coastal waters of the eight main and inhabited islands at the southeastern end of the 2,450 km-long archipelago (Balazs 1982). During recent aerial surveys of Guam (October 1989-April 1991), only 13.2 % of 76 turtles sighted were hawksbills (G. Davis, Guam Div. Aquatic Wildl. Res., in litt., 22 August 1991). In contrast, 83 % of 29 sightings (1971-1991) around Tutuila (American Samoa) were of hawksbills (Tuato'o-Bartley et al. 1992). Elsewhere in the U. S. Pacific specific foraging grounds are known in some cases, such as in Palau (Pritchard 1982, Maragos 1991), but in general the literature for the Caroline Islands, the Republic of the Marshall Islands, and the Commonwealth of the Northern Mariana Islands does not fully differentiate between green turtles and hawksbills (e.g., McCoy 1974, Pritchard 1982, Johannes 1986). There are no documented sightings in the unincorporated U.S. Pacific territories (Groombridge and Luxmoore 1989). However, with the exception of Johnston Atoll (see Balazs et al. 1990), surveys are lacking.

Listing Factors

1. Present or Threatened Destruction, Modification or Curtailment of Habitat or Range

Sea turtles of all species are threatened by the destruction or modification of important nesting and foraging habitats. The number of suitable nesting beaches worldwide has been reduced by residential and commercial coastal development (particularly high-density development), artificial beach-front lighting, uncontrolled recreational use, sand mining, beach armoring (including stabilization structures such as seawalls, groins, and erosion barriers), mechanized beach cleaning, vehicle traffic, litter, the removal of stabilizing dune vegetation, and the introduction of exotic plants. In Barbados, 55.6 % of nests studied were affected by beach lighting, resulting in up to 100 % of newly emerged hawksbill hatchlings in some nests crawling inland (Horrocks et al. 1989). In Antigua, wind erosion following the removal of stabilizing vegetation has rendered significant areas unusable for hawksbill nesting; raking and beach cleaning procedures further exacerbate erosion (Ryder et al. 1989). In Hawaii, resorts, marinas, and even proposed rocket launching facilities (at Ka'u near South Point on the island of Hawaii) threaten hawksbill habitat (NMFS 1992). General overviews of habitat alterations and its observed consequences are provided by Bjorndal

(1982), Groombridge (1982), Coston-Clements and Hoss (1983), Bacon et al. (1984), Hopkins and Richardson (1984), Raymond (1984), National Research Council (1990), and NMFS\FWS (1993).

At Carambola Resort on Davis Beach, St. Croix, 75 hatchlings were recently found dead in runoff basins after having been attracted by beach-front lights and by accent lighting along pedestrian pathways. The basins were later covered with fine-meshed screens, non-essential lights were turned off, and other lights are being replaced with low pressure sodium bulbs to help alleviate the problem. The problem of photopollution is not an isolated one. Hotels along the north coast of St. Croix bathe surrounding cove beaches, virtually all important to hawksbill turtles, in light. On Buck Island, a protected area northeast of St. Croix, researchers patrolling the beach on moonless nights can see their own shadows silhouetted by the bright lights of Christiansted five miles away (Z. Hillis, pers. comm., 1992). Near Frederiksted, artificial lighting leads hatchlings inland and away from the sea (Philbosian 1976). In Puerto Rico, where the lighting problem is also widespread, some cooperation has been obtained from public beaches (e.g., Seven Seas near Fajardo) and private hotels (e.g., Palmas del Mar) with regard to reducing the amount of light shining on nesting beaches at night (B. Pinto, pers. comm., 1992).

At sea, indiscriminate anchoring, chemical and organic pollution, and sedimentation from dredging and upland deforestation degrade coral reefs and other potentially important habitats. In October 1988, the 440-foot cruise ship *Windspirit* illegally dropped anchor west of Francis Bay on the north side of St. John in Virgin Islands National Park and Biosphere Reserve, obliterating 283 square meters of coral reef. In October 1990, the anchor chain of the 438-foot cruise ship *Seabourne Pride* uprooted and overturned at least 42 boulders of living coral (some 3 m across) in 20 m of water off of Caneel Bay, also in the Biosphere Reserve. Furthermore, the cumulative negative effect of countless smaller boats anchoring in coral reef areas is considered significant. Vessel groundings are also a persistent problem and there are many incidents of small boats running aground on shallow reefs off the north shore of St. John (Caroline Rogers, USNPS, pers. comm.). On 15 February 1985, the 350-foot *MV/A. Regina* ran aground off the east coast of Mona Island in federally designated hawksbill Critical Habitat. The wreck spilled diesel oil, extensively damaged the reef, produced a considerable suspension of sediment, and littered the beaches with oil and debris (Cintron and Cintron 1987). On 25 August 1985, a hawksbill emerged to nest on Playa

Sardinera on Mona Island with oil on her fore and rear flippers, plastron, buttocks, cloaca, head and throat; she was unsuccessful in her nesting attempt and did not return (Kontos 1985).

The recent reality of repeated oil spills in the U. S. Caribbean indicates the very serious nature of this threat. In September 1989, following Hurricane Hugo, a 42,000 gallon spill of #6 fuel oil (heavy crude oil) at the Water and Power Authority facility in Christiansted, St. Croix, left south coast beaches heavily oiled. Pelican Cove, a hawksbill nesting beach, was buried under 0.3 m of crude oil. Between March 1991 and March 1992, two more spills, both outside of U. S. waters, threatened U. S. nesting beaches. The first occurred on 6 March 1991, 13 nm north of Nevis, when the Trinidad-registered barge Vestabella, loaded with about 560,000 gallons of #6 fuel oil, sank in 600 m of water after a towing cable snapped; the initial oil slick was more than 30 miles long (Simmonds 1991). According to *The Daily News* (30 March 1991), a USVI newspaper, tar balls and tar sheets began appearing on St. John on 21 March; tar balls washed ashore soon thereafter on St. Thomas, St. Croix, Culebra, Vieques, and the main island of Puerto Rico. One year later, on 15 March 1992, a pipe ruptured during ship-to-shore pumping of #6 fuel oil to a transfer station at St. Eustatius Terminal on the west coast of St. Eustatius, Netherlands Antilles. One hundred barrels of crude oil were released to the sea in a slick that headed northwest out across the rich fishing grounds of the Saba Bank. Heavy seas broke up the slick before it entered U. S. waters, but tar balls eventually fouled the coast of Puerto Rico (Z. Hillis, pers. comm., 1992).

Behavioral experiments indicate that green and loggerhead sea turtles possess limited ability to avoid oil slicks, and physiological experiments show that the respiration, skin, some aspects of blood chemistry and composition, and salt gland function of 15-18 month old loggerheads are significantly affected by exposure to crude oil preweathered for 48 hours (Vargo et al. 1986). There is some evidence to suggest that hawksbills are also vulnerable to oil pollution. Hawksbills (predominantly juveniles), were only 2.2 % (34/1551) of the total sea turtle strandings in Florida between 1980-1984, yet comprised 28.0 % of petroleum-related strandings. Oil and tar fouling was both external and internal. Chemical analysis of internal organs provided clear evidence that crude oil from tanker discharge had been ingested (Vargo et al. 1986). Carr (1987) reported juvenile hawksbills (to 20 cm) "stranded [in Florida] with tar smeared sargassum"; some individuals had ingested tar. He

noted that the Gulf Stream at times carries oil from both European sources and the Gulf of Mexico into Florida waters. More recently, following the Vestabella barge spill, a hawksbill soaked in oil was found dead near Guayama on the south coast of Puerto Rico (B. Pinto, pers. comm., 1992).

2. Over-utilization for Commercial, Recreational, Scientific, and Educational Purposes

Hawksbills were heavily exploited in the United States prior to their federal protection under the Endangered Species Act (ESA) of 1973, as amended. Pre-ESA statistics pertaining to the harvest and marketing of sea turtles in Florida and other southeastern states are provided by Rebel (1974). According to Joe LaPlace (pers. comm. *in* Eckert 1992), a lifelong resident of St. Thomas, the intense commercial harvest of hawksbills began in the USVI in the 1920's when dealers offered to purchase the shells for export. Within just a few years there were fewer turtles in the water and carcasses littered the beaches. After WWII, with the advent of plastics and nylon, the lucrative export market died out, but an increasing human population (residents and tourists) kept the pressure on remaining stocks until protection was conferred in 1970. Today, despite legal protection and depleted local stocks, a black market persists. In St. Croix, meat and eggs are in "high demand" in some areas, with hawksbill and green turtle eggs selling (in season) for \$7-10 per dozen and some "being traded for crack [cocaine]" (Greg Hughes, USFWS, *in litt.*, 12 May 1992). In November 1990, a man in possession of 140 hawksbill eggs was arrested at Manchineel Beach, St. Croix, and convicted in December 1991 after pleading guilty. Another arrest was made in November 1991 after 45 eggs were taken from Sandy Point NWR; the poacher was sentenced to three months in prison and two years probation.

There is also a significant level of clandestine harvest in Puerto Rico, especially on the main island and Mona Island, for both meat and eggs. Seven carcasses were found on Mona in 1986 (Kontos 1987) and four more in 1987 (Kontos 1988). In some western areas of Puerto Rico "hawksbill sausages" are made. The fresh blood is spiced and put in ice trays and the coagulated blood is later fried in hot oil. The "sausages" are a delicacy widely consumed by residents and visitors alike (Pinto 1992). Also on the main island, two men were arrested in August 1987 and their vessel, scuba gear, and 20 pounds of hawksbill meat seized; they were subsequently fined \$2,000 for the take and possession of the meat. In June 1988, one of these men was again convicted of possession, this time of 28.5 pounds of hawksbill meat;

his vessel, car, and fishing equipment were seized and he was sentenced to one year supervised probation and one month in jail. In June 1989, three men pled guilty to counts of taking and possessing endangered species based on their August 1988 arrest for possession of 150-200 pounds of green and hawksbill sea turtle meat; forfeiture of a large vessel employed by the men and all gear is pending (M. Christian, *in litt.*, 31 March 1992). At Cayo Berberia (a mangrove islet off the south coast of Puerto Rico at Santa Isabel), the remains of more than 50 slaughtered hawksbills were found during a site inspection in December 1989 (Vicente et al. 1989).

Illegal harvest for local consumption is also reported from Hawaii and the U.S. Pacific territories. An unquantified number of Hawaiian hawksbills are taken for domestic black markets with spears, nets, harpoons, grappling hooks, firearms from shore, underwater 'bang sticks', nooses, and by hand capture (NMFS 1992). In October 1989, a 212-foot purse seiner docked in Guam was seized after U.S. Department of Commerce officials discovered three small hawksbills aboard the vessel. A fine of \$35,000 was deposited in the Wildlife Conservation Fund as a precondition for release of the vessel (Government of Guam 1990). In recent interviews in American Samoa, respondents in 24 % of the villages reported that turtles (hawksbills and/or green turtles) no longer occurred on their village beaches that historically had nesting activity. During these interviews it was learned that if a sea turtle of any age is encountered, the likelihood of it being harvested is high (Tuato'o-Bartley et al. 1992). Similarly, the take of hawksbills in Palau is unrelenting and residents concede to noticeable declines in recent decades (Johannes 1986, Maragos 1991). Egg theft is also widespread in the U. S. Pacific, except perhaps in Hawaii where hawksbill nesting is relatively rare. In Palau, egg theft claims > 75 % of all nests (Maragos 1991). In American Samoa, villagers will collect eggs for consumption "whenever a nest is found" (Tuato'o-Bartley et al. 1992).

Continuing exploitation in the U.S. is only a small part of a much larger reality. Domestic harvest, often illegal if not covert, is a persistent problem for hawksbills throughout their range. In the Caribbean, hawksbill shell fetched US\$ 110-130/kg in 1980 and a fisherman could earn \$200 or more for a single turtle (Carr and Meylan 1980). The harvest has not abated with the increasing scarcity of the resource; rather, hawksbills are taken opportunistically by divers targeting lobster and other high priced items. Thus it is likely that individuals will continue to be harvested long after the species would otherwise be

commercially extinct. For many coastal Caribbean peoples, the capture and sale of a single hawksbill can be equivalent to one or two weeks' wages. Nietschmann (1981) estimated that 1,000-1,200 hawksbills were removed annually from eastern Nicaraguan waters by nets, harpoons, and hoopings [nooses]; in the decade between 1969 and 1978, the price for shell rose 600 %. Heavy pressure is also exerted on eggs. In Mexico, poachers take more than half of the nests in Campeche and Yucatan, and 60-70 % of those in Isla Holbox; > 50 % of the eggs (and turtles) arriving at the Pacific coast are taken to regional or national markets (PESCA 1990). [N.B. The extent to which the recent ban on sea turtle harvest in Mexico (Aridjis 1990) will ultimately eliminate this market remains unknown]. The scenario is repeated throughout the world (Carr et al. 1982, Meylan 1984b, Groombridge and Luxmoore 1989, Eckert 1993).

Above and beyond domestic harvest is the ongoing international commerce in hawksbill shell (tortoiseshell, or 'bekko'), a phenomenon widely touted as the single most significant factor endangering hawksbill populations around the world. To meet the demands of at least four separate native industries, Japan has conducted the world's largest international trade in sea turtles and sea turtle products, focusing not only on the hawksbill, but on green turtles and olive ridleys (*Lepidochelys olivacea*) as well (Milliken and Tokunaga 1987). Japanese imports of raw bekko between 1970 and 1989 totalled 713,850 kg, representing > 670,000 turtles; more than half the imports originated from the Caribbean and Latin America (Milliken and Tokunaga 1987, updated by Greenpeace to 1989). Between 1970 and 1987, a total of 675,247 kg of stuffed hawksbills representing > 587,000 turtles was imported (Greenpeace 1989). Milliken and Tokunaga (1987) note that in order to maintain these levels of importation, the annual slaughter of at least 28,000 hawksbills is required. Between 1970 and June 1989, Japan imported 368,318 kg of bekko from the Wider Caribbean alone, the equivalent of more than a quarter million turtles; in 1988, Japan imported from the Wider Caribbean the tortoiseshell from nearly 12,000 adult hawksbills (Canin 1989).

In spite of full domestic protection for sea turtles and a ban on their importation into the U. S., sea turtle products, many of them hawksbill in origin, are not only the most commonly confiscated products by U.S. Customs at our borders (Andrea Gaski, TRAFFIC(USA), pers. comm.), but these confiscations are on the increase (O'Connell 1990). And lest we believe that these products are only purchased in foreign countries, about \$150 worth of tortoiseshell jewelry (allegedly imported from Jamaica) was recently confiscated from a store at the St.

Croix airport (G. Hughes, *in litt.*, 12 May 1992). In Puerto Rico, tortoiseshell jewelry valued at approximately \$200 was recently seized by NMFS from a gift shop in La Parguera; a federal investigation determined that the jewelry had been imported by a distributor from Colombia and purchased at a jewelry show in San Juan. The items have been forfeited to the U. S. Government (M. Christian, *in litt.*, 31 March 1992). In 1984, investigations in the USVI resulted in seizure of two commercial shipments of sea turtle jewelry (valued at \$500); in 1986, the seizure of 43 pieces of jewelry; in 1988, forfeiture of \$150 in tortoiseshell jewelry boxes (M. Christian, pers. comm.). It is abundantly clear that the Caribbean tourist industry is fueling the continued exploitation of hawksbills throughout the region for jewelry, trinkets, stuffed turtle wall hangings, and polished shells.

The negative effects of utilizing hawksbill turtles for scientific purposes are not likely to be significant. Contemporary hawksbill research consists primarily of studies of the distribution, abundance, and behavior of nesting and foraging turtles, nest protection, measuring and tagging, biotelemetry, non-intrusive physiological measurements, and other pursuits that are not detrimental to the animals involved. In the United States, the use or take of hawksbills for scientific or educational purposes is controlled by a permit system designed to protect endangered and threatened species.

3. Disease or Predation

There are no data on the extent to which disease affects hawksbill turtles in the wild. External parasites and commensals representing five phyla and 17 families of plants, and two phyla (Arthropoda and Mollusca) and six families of animals have been recorded; dozens of species of internal parasites, mostly trematodes, are also documented (see Witzell 1983).

Ghost crabs (*Ocypode quadratus*, *O. ceratophthalmus*), land crabs (*Cardisoma guanhumi*), monitor lizards (*Varanus indicus*), *Ameiva* lizards, night herons (*Nyctanassa violacea*), and barn owls (*Tyto alba*) are known to prey on eggs and/or hatchlings on the beach (Diamond 1976, Small 1982, Limpus et al. 1983, Márquez 1990, Pinto 1992); the black rat (*Rattus rattus*) is a suspected predator (Hillis and Mackay 1989). Exotic and feral species, such as mongooses (*Herpestes auropunctatus*), wild pigs (*Sus scrofa*), and domestic dogs are also a significant threat in some areas. On St. John (USVI) mongooses destroyed 23 % of the total 1980-1981 egg production, and in 1980 dogs destroyed an additional 19 %; wire enclosures

subsequently helped to mitigate this problem (Small 1982). On Buck Island, north of St. Croix, mongooses destroyed virtually all nests laid (at least 45) in 1981 (Small 1982) but subsequent eradication programs reduced the mongoose population and in 1988 only 8 % of nests there were lost to this predator (Hillis and Mackay 1989). On Mona Island in 1987, 35 of 71 (49 %) nests were lost to feral pigs (Kontos 1988). Richardson (1990) reports 50-100 % annual nest loss to pigs on some of Mona's most important hawksbill beaches. After galvanized fences were installed in June 1990, pig predation ceased (B. Pinto, pers. comm., 1992).

At sea, hatchlings, juveniles, and adults fall prey to oceanic predators, including tiger sharks (*Galeocerdo arcticus* [= *cuvier*]), groupers (*Epinephelus itajara*), and crocodiles (*Crocodylus porosus*) (summarized by Witzell 1983). Hatchlings were attacked by a black-tipped reef shark (*Carcharhinus spallanzani*) in Samoa (Witzell and Banner 1980). In 1982 the carapace scutes from an estimated 28 kg hawksbill were found in the stomach of a 4 m tiger shark captured off St. Thomas, USVI (Boulon 1984).

4. Inadequacy of Existing Regulatory Mechanisms

In the United States, both national and international laws protect hawksbill turtles from harassment, harvest, and commerce. According to the Code of Federal Regulations (50 CFR 17.11), the hawksbill is listed as Endangered throughout its entire range under the U. S. Endangered Species Act (ESA) of 1973, as amended (35 FR 8495; June 2, 1970). Similarly, the species is classified as Endangered in the International Union for Conservation of Nature and Natural Resources' (IUCN) *Red Data Book*, where taxa so classified are considered to be "in danger of extinction and whose survival is unlikely if the causal factors continue operating" (Groombridge 1982). Hawksbills are also included on Appendix I of the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES). The U. S. ratified CITES in 1974.

All federal agencies must ensure that actions authorized, funded, or carried out by them do not result in the destruction or adverse modification of habitat designated as Critical Habitat for marine turtles pursuant to Section 7 of the ESA. Critical Habitat was proposed (Federal Register, 24 May 1978; 43 FR 22224-22225) for hawksbill turtles in Puerto Rico by the U. S. Fish and Wildlife Service and subsequently designated (effective 26 July 1982) as follows:

(a) Mona Island. All areas of beachfront on the west, south, and east sides of the island from mean high tide (MHT) inland to a point 0.1 miles from shore, including all 7.2 km of beaches. (b) Culebra Island. All areas of beachfront on the north shore of the island from MHT inland to a point 0.1 miles from shore, including the following beaches: Playas Blanca, Flamenco, Resaca, Brava, and Larga. (c) Cayo Norte. South beach, from MHT inland to a point 0.1 miles from shore. (d) Culebrita Island. All beachfront areas on the southwest, east, and northwest shores of the island from MHT inland to a point 0.1 miles from shore (Dodd 1978).

Critical Habitat for this species was also proposed elsewhere in Puerto Rico (Vieques Island, south beaches of Culebra Island), as well as in the USVI (portions of St. Croix, St. John, and St. Thomas; beaches of Buck Island National Monument) (Dodd 1978), but these proposals have not been acted upon (Ken Dodd, USFWS, pers. comm., 1992). Critical Habitat has not been designated for hawksbills (or for any other species of sea turtle) in the U. S. Pacific. A proposed rule was prepared in 1980 to designate Critical Habitat for the green turtle in Hawaii, Republic of the Marshall Islands, Federated States of Micronesia, and the Republic of Palau, but this was never approved by the U. S. Fish and Wildlife Service (K. Dodd, pers. comm.). In addition, several beaches were considered by Dodd (1978) to be potential candidates for Critical Habitat for the hawksbill turtle in the U. S. Pacific, including American Samoa (Tutuila Island, Rose Atoll, Swains Island) and portions of the Republic of Palau, Truk District, and Lower Mortlocks. The protection of important habitat is crucial to the long-term survival of endangered and threatened sea turtles. In light of the fact that no progress whatsoever has been made during the last decade with regard to the designation of Critical Habitat for hawksbill sea turtles under the ESA, the evaluation and designation of such habitat should be a high priority.

Broad legal protection has been effective in eliminating turtle and egg poaching in the continental USA, but not so in Puerto Rico, the USVI, or the U.S. Pacific. In American Samoa, for example, most people are unaware that it is illegal to take turtles and enforcement of existing laws protecting turtles is lacking (Tuato'o-Bartley et al. 1992). In Puerto Rico the theft of eggs is nearly ubiquitous and inadequate local law enforcement is a widely recognized problem. Meat is available for the asking on the black market and, in spite of restrictions, some businesses continue to exhibit and sell items fashioned from hawksbill shell (Pinto 1992). Poaching also continues in the USVI. The Virgin Islands Department of

Planning and Natural Resources' Law Enforcement Division lacks sufficient funds and manpower to fully cover the beaches and phones (at least on St. Croix) are unmanned on weekends when most people observe violations (Z. Hillis, *in litt.*, 11 May 1992). The enforcement problem is only exacerbated by the diffuse distribution of this species, by some nesting and foraging habitats being coincident with relatively remote territories, such as Mona Island in the Caribbean and the varied archipelagoes of the Pacific, and by the inexperience or inattention of Customs officers that permits products derived from endangered species to pass into and out of U.S. Caribbean and Pacific islands. Vigilant enforcement and more visible public awareness campaigns are clearly needed.

Elsewhere in the world, despite the protection conferred by CITES, harvest continues for several reasons. First, while CITES prohibits trade in sea turtle products amongst nations, it does not restrict domestic harvest or commerce. Second, while 126 nations have ratified CITES (WWF/IUCN 1994), some have exercised their right to take exemption to treaty provisions as they pertain to sea turtles. When Japan ratified CITES in 1980, it placed reservations on the hawksbill, green, and olive ridley, effectively exempting itself from the ban on their trade (Greenpeace 1989). [N.B. Japan has since withdrawn all reservations on sea turtle species]. Similarly, Cuba took (and still maintains) reservations on hawksbills and green turtles when it acceded to the Convention in 1990. Third, some countries, such as Indonesia which ratified CITES in 1979, ignore their obligations as CITES parties and openly trade in Appendix I species. According to Japanese Customs Statistics, stuffed hawksbills from Indonesia accounted for nearly half of all "worked bekko" imports between 1979-1986 (208,586 kg out of 440,914 total kg; Milliken and Tokunaga 1987). Other CITES parties clandestinely participate in the trade by re-routing their shipments through a non-CITES country. Finally, some important hawksbill suppliers do not belong to CITES at all. Between 1970 and June 1989, Haiti, a non-CITES country, exported bekko representing > 21,000 hawksbills to Japan (Greenpeace 1989).

5. Other Natural or Manmade Factors

On the beach, eggs can be lost to flood waters, inundation, and erosion (Small 1982, Hillis and Mackay 1989, Horrocks et al. 1989, Hillis et al. 1990). In 1989, 18 % of hawksbill nests at Buck Island Reef National Monument (St. Croix) were lost to flood tides and erosion effected by Hurricane Hugo. During 1990, there was a three-fold increase in the false crawl:

nest ratio on Buck Island due to steep erosion berms, downed trees, and exposed root masses which remained as Hugo's legacy (Hillis 1991). The problem persisted in 1991, as fallen vegetation continued to interfere with nesting attempts in the beach forest (Hillis 1992). In Hawaii, an entire nesting beach at Harry K. Brown Beach Park in Kalapana was recently lost to a lava flow from the Kilauea volcano (NMFS 1992).

In nearshore waters, hawksbills are periodically captured in the cooling water intakes of industrial facilities, such as Florida Power and Light Company's St. Lucie Power Plant on Hutchinson Island. Between March 1976 (when the St. Lucie Plant opened) and November 1988, six hawksbill captures occurred in the Plant (Ernest et al. 1989). An additional three were recorded as of 1 June 1992, the most recent being a female weighing 191 lb [87 kg] and captured on 1 March 1991; all turtles were released unharmed (Eric Martin, Applied Biology, pers. comm.). In offshore waters, such as in the Gulf of Mexico, oil and gas structures act as "artificial reef habitats" and the explosive removal of these structures may result in the death of turtles (Gitschlag and Renaud 1989).

Incidental capture and entanglement at sea are unquantified and potentially important sources of mortality. The Sea Turtle Stranding and Salvage Network (Atlantic and Gulf coasts of the U. S.) reported 27 stranded hawksbills in 1988, 35 in 1989, 61 in 1990 (when an unusually large number of post-hatchlings washed ashore in Texas), 33 in 1991 and 47 in 1992 (Wendy Teas, NMFS, pers. comm.). While hawksbills typically represent < 1.5 % of total sea turtle strandings (Schroeder and Warner 1988, Teas and Martinez 1989), the problem is a persistent and worrisome one. Gill nets, longlines, and shrimp trawls all take turtles in Gulf of Mexico waters (Hildebrand 1987, NRC 1990) and hawksbills strand on the Texas coast during virtually all months of the year (Schroeder and Warner 1988, Amos 1989, Teas and Martinez 1989, 1992).

Of the 25 sea turtles found entangled on the Texas coast during 1986 and 1987, 24% were juvenile hawksbills (mean 24.3 cm CCL) (Plotkin and Amos 1988). Hawksbills (predominantly juveniles) have been reported entangled in monofilament gill nets, "fish nets", fishing line, and synthetic rope; in most cases flippers were lost as a result, and in one case an animal was recovered with a piece of plastic onion bag entangled around its neck (Balazs 1985). Balazs (1985) summarized published incidents of the ingestion of marine

debris by hawksbills and reported that 88.9 % of the articles recovered were plastic bags, plastic and styrofoam particles, and tar; 90.9 % of the individuals involved were juveniles.

The specific effects of marine debris and pollution, including oil and pesticides, on hawksbills, their eggs, and their prey have yet to be determined. The ability of hawkbill populations to recover will ultimately depend not only on protection from direct harvest and the conservation of important nesting beaches, but also on clean oceans and a healthy littoral zone, especially as this pertains to coral reefs and other foraging and resting grounds.

Conclusion

The hawkbill is listed as Endangered by the U. S. Endangered Species Act of 1973, as amended (USFWS 1989) and the IUCN *Red Data Book* of endangered and threatened species (Groombridge 1982). It is also included on Appendix I of CITES. With the exception of the Kemp's ridley (*Lepidochelys kempii*), the hawkbill is considered by many to be the most endangered of all the marine turtles. Of 65 geopolitical units where estimates of relative hawkbill nesting density exist, 38 of them have hawkbill populations that are suspected or known to be in decline and an additional 18 have experienced "well-substantiated declines" (Groombridge and Luxmoore 1989). Hawksbills are a shared resource in the Western Atlantic, meaning that the status of U.S. populations is tied to factors operating outside our jurisdiction. Small juveniles in Texas waters are likely to originate from the nesting beaches on Mexico's Yucatan Peninsula (Hildebrand 1987, Amos 1989) and perhaps other Caribbean beaches as well. Tagging studies have shown that juveniles tagged in St. Thomas are frequently recaptured outside of USVI waters, including locales as distant as St. Lucia 650 km to the southeast (Boulon 1989). The same is true in the U.S. Pacific, where hawkbill stocks are likely to be shared in common amongst the islands of the central and western Pacific.

The greatest threat to the hawkbill worldwide is commerce in raw and worked shell ('bekko', or tortoiseshell). The advent of plastics in the 1940s seemed to promise a more secure future for the species, but this has not been so. Until recently, tens of thousands of hawksbills were sacrificed each year to meet the demand for jewelry, ornamentation, and whole stuffed turtles (Milliken and Tokunaga 1987). Hawksbills are mobile, and thus international trade threatens not only the stocks of exporting countries but all hawkbill

populations, including those of the U.S.. Japan's 1988 bekko imports from Jamaica, Haiti, and Cuba represented some 13,383 hawksbills; it is extremely unlikely that this volume could have originated solely from local waters (Greenpeace 1989).

In a move destined to benefit hawksbill populations in the U.S. and throughout the world, Japan announced on 19 June 1991 its intention to end its international trade in bekko. The decision followed a determination by the U.S. Government that Japan was "undermining the effectiveness of international programs for the conservation of sea turtles", a determination that, under the authority of the Pelly Amendment to the Fisherman's Protective Act of 1967, carried the threat of a partial or complete embargo on fish and wildlife products imported into the U.S. from Japan (Donnelly 1991). Japan kept its promise to ban all imports of turtle shell by 31 December 1992, and subsequently removed its *Eretmochelys* reservation under CITES in 1994.

There is little doubt that U.S. Caribbean nesting colonies have been severely depleted during the twentieth century. The same can be said for the Pacific, where the advent of modern fishing gear has made harvesting turtles more efficient, and the means to travel with motorized vessels safely over great distances has enabled hunters to gather mating and nesting turtles at previously inaccessible breeding sites. Today the illegal domestic harvest of eggs and turtles continues in the United States, especially in Caribbean and Pacific island territories. Law enforcement, as well as conservation and management efforts, are hindered by diffuse nesting distributions and the remoteness of some rookeries. It is not easy to determine whether remaining populations are stable, increasing, or declining. Systematic surveys of selected index beaches are essential in order to evaluate the present status of remnant stocks. Despite predictions that hawksbills would be extinct on Mona Island before the year 2000 (Kontos 1988), there was cautious optimism after seven years of study (1974, 1984-1989) that annual fluctuations in nesting activity were not projecting a long-term decline (Richardson 1990).

While some areas under U.S. jurisdiction, notably Mona Island, St. John, and Buck Island in the Caribbean and Palau in the Pacific, have the potential to serve as important refugia for the species, the degradation and loss of habitat continues in the United States and elsewhere. Coastal development, including beach armoring, traffic, and artificial lighting is proceeding at a rapid pace. Coral reefs, vital as food and shelter for hawksbills, are shattered by

indiscriminate anchoring, broken by divers, fouled by pollutants and sedimentation, stressed by disease, and smothered by spilled oil. At-sea entanglement in active and abandoned fishing gear, the ingestion of marine debris, and petroleum fouling are unquantified and potentially serious concerns to the turtles themselves. In summary, depleted U.S. populations are not currently considered to be declining, but neither are there indications of recovery despite more than a decade of legal protection. In addition, habitat destruction and clear evidence of over-exploitation continue on a global scale. The best available commercial and scientific data indicate that the hawksbill turtle (*Eretmochelys imbricata*) should remain listed as Endangered throughout its range, pursuant to Section 4 of the Endangered Species Act of 1973, as amended.

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Kemp's Ridley Sea Turtle, *Lepidochelys kempii*

Prepared by Michael Weber

Biological Background

Kemp's ridley sea turtle has also been known as the Atlantic ridley, Mexican ridley, bastard turtle, or grey loggerhead in English; as tortuga lora or tortuga cotorra in Spanish; and as tortue batarde in French. The specific name derives from Mr. Richard Kemp of Key West, Florida, who provided the first type to Garman, who then described the species. The name "ridley" replaced "bastard turtle" several decades ago, reflecting common usage along the U.S. Gulf of Mexico coast where the animal was once common. The derivation of the word ridley for sea turtles in the Genus *Lepidochelys* is unclear.

Kemp's ridley exhibits several distinguishing features: two pairs of prefrontal scales, five (very rarely more) pairs of costal scutes, a pore in each inframarginal scute in the bridge, paddle-shaped forelimbs, each with one claw, and a shell, in adults, nearly as wide as long (Mager 1985, Pritchard 1989, Ross et al. 1989). Kemp's ridley is one of the smallest of all marine turtles. Adult females measure 58 - 80 cm in straight carapace length (SCL), and weigh 40 - 50 kg (Ross et al. 1989). The tail of an adult male extends well beyond the carapace; the tail of an adult female does not (NRC 1990). Juvenile and hatchling male and female Kemp's ridleys are morphologically indistinguishable. Hatchlings are roughly 4.1 cm in carapace length. Compared to the olive ridley (*Lepidochelys olivacea*), Kemp's ridley is similar in size and general conformation, but has smaller orbits in a broader head with more massive, powerful jaws (Pritchard 1989, NRC 1990). The carapace coloration of Kemp's ridley appears light olive to light gray in adults; charcoal gray in juveniles, and dark gray to black in hatchlings. In juvenile Kemp's ridleys, the carapace can be wider than it is long (Pritchard 1989). The plastron or ventral shell appears white or cream-colored in adults and juveniles, and usually the same color as the dark gray carapace or a slightly lighter shade in hatchlings.

It is unknown for certain how long Kemp's ridleys live, or how long it takes them to reach sexual maturity in the wild. Recent estimates using skeletochronological data fitted to a Von Bertalanffy growth curve suggest that female Kemp's ridleys may attain the size of a sexually

mature individual within 11 - 12 years (Zug 1990). In captivity and on a steady, high protein diet, Kemp's ridleys as young as five to seven years have produced eggs and hatchlings (Wood and Wood 1988).

Kemp's ridleys are distributed primarily in the Gulf of Mexico and western Atlantic Ocean. Kemp's ridleys may also occur in the eastern Atlantic Ocean, however most documented occurrences have been juvenile individuals (Bleakney 1965, Brongersma 1972, Pritchard and Marquez 1973). Unlike most sea turtles, Kemp's ridley has a very restricted nesting range. The principal Kemp's ridley nesting beach is located near Rancho Nuevo, Tamaulipas, Mexico where 95 % of the nests are laid along 60 km of beach. Hurricane Gilbert scoured the beach in 1988, removing the foredune over a long stretch of beach and leaving a mixture of gravel, sand, and rock rubble. In 1989, females apparently expanded their nesting up to 30 km northward of the traditional nesting beach (NRC 1990).

Nesting also occurs sporadically at other beaches in Mexico and in Texas (Ross et al. 1989), and typically involves only one or very few individual turtles. There have been a few instances of Kemp's ridleys nesting on beaches outside their traditional range in recent years. Sightings of female Kemp's ridleys emerging on beaches, their tracks, and/or their eggs and hatchlings have been documented on Florida's Gulf and Atlantic coasts (Meylan et al. 1990) and on the Atlantic coast of North Carolina (T.A. Conant unpublished data cited in Bowen et al. 1994) and South Carolina (Anonymous 1992).

Unlike sea turtles of other genera, Kemp's ridleys emerge synchronously during the day to nest in aggregations called "arribadas" meaning "arrival" in Spanish. Strong onshore winds appear to stimulate nesting emergence and to concentrate nesting effort in a narrower area than on calm days (Pritchard 1989). More than half of the adult females nest every year between April and mid-August, while the remainder may or may not skip certain years (NRC 1990). Kemp's ridleys lay on average 3.075 clutches per season (Rostal 1991), at 20 - 28 day intervals, with an average of 103 eggs per clutch. The nesting process takes less than an hour (Chavez 1969).

After a 48 - 65 day incubation period, viable eggs hatch and the hatchlings emerge about dawn (Mager 1985). After leaving their nests, hatchlings swim determinedly at the surface, eventually encountering overwhelming currents, such as the Loop Current, which carry them

elsewhere in the Gulf of Mexico or into Atlantic waters where they presumably inhabit *Sargassum* drift lines, convergences, eddies, and rings where they feed at the surface (Carr 1986, 1987) until reaching a carapace length of about 20 cm (NRC 1990). Post-pelagic stage Kemp's ridleys measuring about 20 - 25 cm carapace length become benthic feeders when they enter inshore and nearshore waters, particularly areas where there are seagrass beds or mud bottoms favored by crabs, from Long Island Sound to the northern Gulf of Mexico. Concentrations of juvenile ridleys measuring 20 - 58 cm carapace length are documented for Cape Cod Bay, Long Island Sound, Chesapeake Bay, and bays and sounds of North Carolina, northwest Florida, western Louisiana, and along the Texas Gulf coast (Keinath et al. 1987, Ross et al. 1989, Morreale and Standora 1991, Rudloe et al. 1991, Manzella and Williams 1992, Kenyon et al. 1994, Landry et al. 1994, Epperly et al. 1995).

Along the U.S. Atlantic coast, juvenile ridleys inhabit bays and sounds from Spring until Fall when most turtles depart inshore areas and move south as water temperatures decline (Keinath et al. 1987, Byles 1988, Morreale and Standora 1993). For reasons unknown, some Kemp's ridleys remain in inshore waters during Winter and are later found cold-stunned after water temperatures decrease to near freezing temperatures (Morreale and Standora 1993). Juvenile Kemp's ridleys in the Gulf of Mexico occupy inshore waters and nearshore waters from shore to approximately 50 m in depth (Rudloe et al. 1991, Shaver 1991, Renaud 1993). Juvenile Kemp's ridleys move from shallower to deeper waters as temperatures cool during Winter (Henwood and Ogren 1987).

Within the Gulf of Mexico, adult Kemp's ridleys occupy nearshore waters from shore to approximately 50 m in depth (Byles 1988, Shaver 1991). Post-nesting adults tracked by satellite from Rancho Nuevo, Mexico migrated along narrow corridors in coastal waters generally less than 50 m deep and swam to feeding areas where they established circumscribed ranges (Byles 1988). These female ridleys moved as far south as Cabo Catoche at the northeast point of the Yucatan Peninsula, where the Gulf Loop Current may have discouraged further movement, and as far north as northwestern Louisiana (Byles 1988). Nothing is known about the migrations of adult male Kemp's ridleys.

When adult Kemp's ridleys are not migrating to or aggregated near their principal nesting beach, they inhabit crab-rich waters, such as those close to the Mississippi River Delta and Laguna del Carmen in the Bay of Campeche (Pritchard 1989, NRC 1990). Adult Kemp's

ridleys are carnivorous benthic feeders, preferring crabs, but also occasionally eating molluscs, fish, shrimp, and vegetation (Mortimer 1982, Lutcavage and Musick 1985, Shaver 1991, Burke et al. 1993, Werner and Landry 1994). The stomach contents of stranded Kemp's ridleys often include fish remains. Since Kemp's ridleys are not likely able to capture live fish, these stranded animals were probably feeding on dead fish, in many cases those discarded by shrimp trawlers (NRC 1990, Shaver 1991).

Both eggs and hatchlings are vulnerable to a wide array of predators, including coyotes, ghost crabs, vultures, caracaras, and hawks. Marine predators of hatchlings include jackfish, red drum, sharks and humans (NRC 1990). Juveniles and adults are vulnerable only to larger predators such as sharks and humans.

Population Size

The best index for population size of Kemp's ridley, like other sea turtles, is the total number of nests and nesting females found on nesting beaches (NRC 1990). Based on a 1947 film by amateur naturalist Andres Herrera of Tampico, Mexico, an estimated 42,000 Kemp's ridleys nested at Rancho Nuevo on June 18, 1947 (Hildebrand 1982). Since that time, the nesting population has drastically declined. By 1966, when the Mexican government initiated protection of the nesting beach, the largest arribadas included no more than 2500 females. Assuming that each nesting female lays 3.075 clutches per season (Rostal 1991), the nesting population is now estimated to be 509 females (Table 1), or a little more than one percent of the nesting population in 1947.

There is no quantitative information on the abundance of adult males and only a little on juveniles. Ogren (1989) has observed that juvenile Kemp's ridleys are now regularly encountered in the northern Gulf of Mexico, whereas surveys in the 1950s had encountered none.

Listing Factors

1. Present or Threatened Destruction, Modification or Curtailment of Habitat or Range

In 1977, the Government of Mexico designated Rancho Nuevo a Natural Reserve. Although development within the reserve remains minimal, land clearing has been intense. There are two additional causes for concern that incompatible development may be in the offing. First, a local group of townspeople, who formed a fishing cooperative, constructed several concrete buildings in the reserve adjacent to the Barra Coma dune and just north of the turtle camp. The cooperative caused minor environmental problems, such as small spills of oil and gas on the ground and careless discard of garbage. Concern about the cooperative has decreased, since the townspeople, inexperienced as fishermen, have not succeeded economically (R. Byles pers. comm.). Second, a road through the reserve from Barra Coma to Barra del Tordo has been proposed. Although construction of the road has been delayed and direct impacts on the nesting beach are unlikely, the proposal suggests that pressure to develop the beach is building.

Point and non-point source discharges of agricultural and industrial chemicals, petroleum products, and domestic sewage may have indirect effects on Kemp's ridleys by reducing food sources through degradation of the habitat of prey species (such as sea grass beds and blue crabs), and direct effects such as reduction in health and fitness of individual animals that may be manifested by mortality or the disruption of physiological functions. Many of the 347 industrial discharges in the U.S. Gulf of Mexico are found along the eastern Texas\western Louisiana coast that is favored by juvenile ridleys (OTA 1987). Similarly, agricultural runoff of nutrients and pesticides, urban runoff of oil, heavy metals, and other substances, and discharge of nutrients from the Mississippi and other rivers may have led to the loss of prey nursery areas. Levels of discharges for much of the Gulf, particularly Mexico, are unknown. Direct effects of these pollutants on Kemp's ridleys have not been thoroughly investigated.

Besides possible indirect effects upon sea turtles of heavy metals and other toxic substances in muds and waters discharged in the course of routine operation of exploratory and production oil and gas platforms, Kemp's ridleys may be exposed to oil released in catastrophic spills from oil rigs and tankers. Evidence suggests that sea turtles do not avoid oil slicks and can die by coating and ingestion of oil (Fontaine 1989).

Chronic release of small amounts of oil from oil rigs and from commercial oil tankers and freighters emptying their bilges creates tarballs that Kemp's ridleys may confuse with food.

Tarballs may cling to the mouth of a turtle preventing it from eating; if ingested, a tarball may clog the esophagus. The toxic components of a tarball can also have fatal consequences.

Nearshore habitats important to Kemp's ridleys are regularly exposed to routine and catastrophic release of oil and other hazardous materials. The U.S. Gulf of Mexico includes six of the top ten ports in the amount of crude oil and refined products handled and in the number of barge movements (Townsend 1990). Mexico's largest offshore oil field, located in the Bay of Campeche, has already been the source of the major blow-out and spill from the Ixtoc I well, and is certainly the source of routine discharges.

Oceanographic features such as convergences and currents concentrate discarded debris, making the Gulf of Mexico particularly hazardous to Kemp's ridleys (O'Hara and Debenham 1989). Debris may affect sea turtles in several ways. Pieces of netting and plastic bags, for instance, may entangle sea turtles, making it difficult for them to swim, feed, digest, or evade predation. Sea turtles also ingest both large and small pieces of debris, sometimes causing reduced nutrient absorption, intestinal damage, or possibly a false feeling of satiation (Ross et al. 1989). Ingested debris is often present, but cannot always be implicated as a cause of death in necropsied animals (Plotkin and Amos 1988).

There is little information on the impacts of dredging and filling on Kemp's ridleys in the Gulf of Mexico. Past dredging of channels has contributed to the loss of coastal wetlands, particularly in Louisiana, that juvenile ridleys favor. The loss of similar developmental habitat has not been quantified for the Atlantic coast.

2. Overutilization for Commercial, Recreational, Scientific and Educational Purposes

Taking of eggs from nests and from slaughtered nesting females together with incidental drowning in shrimp trawls and other fishing gear led to the historical decline of the nesting population. Both Mexican and U.S. law now prohibit intentional taking of Kemp's ridley sea turtles and their eggs. Working cooperatively, the two governments have all but eliminated poaching on the nesting beach (Ross et al. 1989). Incidental capture of Kemp's ridleys in the U.S. shrimp trawl fishery which once killed more Kemp's ridley than all other human activities combined (NRC 1990), continues to threaten the existence of this species despite federal regulations requiring the use of TEDs in all areas, at all times.

3. Disease or Predation

Since the cooperative Mexican \ U.S. recovery program began in 1978, predation on nests has been held to no more than five percent each year. Little is known about mortality caused by disease or parasites, except for animals reared in captivity (Leong et al. 1989).

4. Inadequacy of Existing Regulatory Mechanisms

Deliberate taking of Kemp's ridleys was prohibited by law in the United States and Mexico in 1973 (Marquez 1989). Shrimp trawling in the vicinity of the nesting beach at Rancho Nuevo is also prohibited, but enforcement of this prohibition should be strengthened. The U.S. Endangered Species Act's prohibition on taking of any kind has been vigorously enforced, with the exception of incidental capture in shrimp trawls. Federal regulations now require the use of Turtle Excluder Devices (TED) on all shrimp vessels in the U.S. range of the Kemp's ridley. Enforcement of TED regulations in recent years has not been sufficient enough to keep mortality levels at a minimum. Several hundred juvenile Kemp's ridleys recently stranded along the Texas and Louisiana Gulf coast (Shaver 1994), presumably a result of incorrect installation of TEDs in some shrimp trawl nets. Reducing mortality among juvenile Kemp's ridleys is the key to reversing the decline in numbers of nesting females (NRC 1990).

5. Other Natural of Manmade Factors

Increases in the number of pleasure boats in nearshore and estuarine areas are no doubt leading to increased numbers of collisions with sea turtles. The level of mortality from this human activity is unknown, however it is probably small (NRC 1990).

There is evidence that Kemp's ridleys may be killed in the removal of offshore oil rigs through explosives (Klima et al. 1988). The level of mortality from this activity is unknown, but is probably small. Measures to reduce this source of mortality have been implemented (Richardson 1989).

Kemp's ridleys are also captured in small numbers in other fisheries, such as the Long Island Sound and Chesapeake Bay pound net fisheries (NRC 1990). Powerplant intake pipes entrap

and drown a small number of Kemp's ridleys (NRC 1990). There is potential for incidental killing of Kemp's ridleys in dredge operations throughout the Gulf of Mexico and southeast Atlantic coastal waters. Three juvenile Kemp's ridleys were killed in dredges at the Kings Bay Naval Base in Georgia (Ross et al. 1989).

Conclusion

The Kemp's ridley is the most endangered of all sea turtle species and shows little sign of recovery. A minor increase in nesting may be an artifact of greater nesting beach coverage or may indicate that the nesting population has increased. The lower average clutch size (95 eggs) that has been reported in recent years may indicate an influx of novice nesters (P. Pritchard pers. comm.).

Efforts to restore the species must concentrate upon protecting subadult and adult animals (NRC 1990) and must take place over a long period of time because of the species' slow maturation. The key to this task is the use of TEDs throughout the U.S. and Mexican shrimp fisheries in adult and developmental habitats, and continued protection of the species' nesting beach.

The best available commercial and scientific information indicates that Kemp's ridley is severely depleted and in danger of extinction (Ross et al. 1989, NRC 1990). Therefore, this species should continue to be listed as an endangered species throughout its range.

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Table 1. Annual nest counts at Rancho Nuevo, Mexico and estimated size of the nesting population of Kemp's ridley turtle (*Lepidochelys kempii*) (R. Byles, pers. comm.). (Note: the length of beach surveyed increased from the early years until the present. The area surveyed has held constant since 1987. Number of nesting females is calculated based on an average 3.075 clutches per turtle per season Rostal (1991)).

<u>Year</u>	<u># Nests</u>	<u># Nesting Females</u>
1978	924	301
1979	954	310
1980	868	282
1981	897	292
1982	750	244
1983	746	243
1984	798	260
1985	702	228
1986	744	242
1987	737	240
1988	842	274
1989	878	286
1990	992	323
1991	1155	376
1992	1275	415
1993	1184	385
1994	1566	509

Olive Ridley Sea Turtle, *Lepidochelys olivacea*

Prepared by Peter C. H. Pritchard, Ph.D. and Pamela T. Plotkin, Ph.D.

Biological Background

The olive ridley is a small, hard-shelled marine turtle, one of the two species of the genus *Lepidochelys*, and a member of the family Cheloniidae. The species may be identified by the uniquely high and variable numbers of vertebral and costal scutes. Although some individuals have only five pairs of costals (the number shown by almost all individuals of the congener *Lepidochelys kempii*), in nearly all cases some division of costal scutes occurs, so that as many as six to nine pairs may be present. Division of the "standard" scutes occurs from the rear of the carapace, so that a specimen with, say, seven pairs of costals shows division of the homologs of costals IV and V. Asymmetry in the number of costal scutes is frequent.

In addition to the division of the costal scutes, the vertebral scutes also show frequent division, as do the scales on the dorsal surface of the head. The prefrontal scales, however, typically number two pairs. The carapace is wide in subadults and adults, although less so than that of *L. kempii*. In anterior profile it is typically elevated and flat-topped, with flat, sloping sides. The plastron is large, with the usual six pairs of large scutes and sometimes a small intergular and interanal also. The inframarginals typically number four on each side, each of which is perforated by a pore located towards its posterior margin. The head is relatively large, as compared with that of *Chelonia* or *Eretmochelys*, but is smaller than that of adult *Caretta*, and slightly smaller than that of *L. kempii*.

The skull of *L. olivacea* differs from that of *L. kempii* in many ways. The orbit is consistently larger, the width across the pterygoids is consistently less, and the alveolar ridges are sharp and are only evident on the rhamphothecae, not on the underlying maxillary-palatine sutural area. There are also numerous differences between the lower jaws of the two species, as follows: i) the entire bony alveolar surface of *L. olivacea* is flat (rather strongly concave in *L. kempii*) ii) the overall depth of the jaw of *L. olivacea* is somewhat less than that of *L. kempii* iii) the tip of the coronoid bone, that provides attachment for the temporal muscle closing the jaw, is rounded off in *L. olivacea* (bluntly pointed in *L. kempii*)

iv) the dorsal mandibular vacuity is relatively larger in *L. olivacea* than in *L. kempii* v) the articular surfaces of the mandible of *L. olivacea* tend to be directed upward (directed more posteriorly in *L. kempii*) vi) the bones comprising the articular surfaces (the articular, pre-articular, and angular) are loosely sutured in adult *L. olivacea*.

In size, female olive ridleys in Surinam range in carapace length from 62 to 74 cm (mode 69 cm); in Pacific Honduras from 58 to 74 cm (mode 65 cm); and in Pacific Mexico from 56 to 78 cm (mode 68 cm) (Pritchard 1969). Márquez et al. (1976) reported upon the carapace lengths of thirteen adult male olive ridleys from Baja California. They ranged from 58.5 to 69.0 cm.

The range of the olive ridley is essentially tropical. In the eastern Pacific nesting takes place from southern Sonora, Mexico, south at least to Colombia. Non-nesting individuals occasionally are found in waters of the southwestern United States (California), whilst they occur abundantly in Pacific Colombia and Ecuador, but only in small numbers in Peru and Chile.

The olive ridley has been recorded occasionally from Galapagos waters, but it is essentially very rare throughout the islands of the Pacific, and indeed even in the western Pacific it is scarce everywhere, although widespread low-density nesting occurs. In the Indian Ocean it only achieves abundance in eastern India and Sri Lanka, although minor nesting occurs alongside the green turtles at Hawke's Bay, Pakistan, and some nesting also occurs in New Britain, Mozambique, Madagascar, peninsular Malaysia, and various other localities.

In the Atlantic Ocean, the olive ridley occurs widely, but probably not in great abundance, in waters of West Africa, from about Mauritania southward at least to the Congo. In the western Atlantic, nesting formerly occurred abundantly in eastern Surinam (Eilanti and Bigisanti Beaches), as well as in western French Guiana and northwestern Guyana. Non-nesting individuals occur regularly as far west as Isla Margarita and Trinidad, but they rarely penetrate any further into the Caribbean than this. The species occurs in Brazil, and nests in the states of Bahia and Sergipe (G. Marcovaldi, pers. comm. to H. Reichart), but it seems to be rare.

Geographic variation in olive ridleys is subtle, and no subspecies are currently recognized. However, the number of costal scutes apparently varies from one area to another (Pritchard 1969), specimens with only five pairs of costals being somewhat more abundant in the eastern Pacific than elsewhere. In addition, overall carapace coloration is typically somewhat lighter in the western Atlantic than in the eastern Pacific. and the shell is typically more elevated in the eastern Pacific than elsewhere.

The olive ridley was formerly known as the Pacific Ridley, until it became generally recognized that the species also occurred in tropical parts of the Atlantic Ocean also (as well as the Indian Ocean). In Mexico it is called "golfina" or "tortuga bestia," whereas in Pacific Costa Rica it is usually called "lora" or "carpintera." From Guatemala to Nicaragua it is called "paslama". In Trinidad it is called "batali"; in Guyana, "terecay"; and in Surinam and French Guiana, "warana". Malay names include "penyu lipas", "penyu rantau," and "penyu abu abu". In India, the Bengali name is "samudrik katha"; the Hindi name is "gahda kachua"; the Oriya name is "samudrik kachchima;" the Tamil is "pul amai," "sith amai," or "kadai amai," and the Telegu name is "samudram thabelu."

The most dramatic aspect of the life history of the olive ridley is the habit of forming great nesting aggregations, generally known as "arribadas," sometimes as "arribazones" or "morriñas". Although not every adult olive ridley participates in these arribadas, the vast majority of them do. Formerly these nesting concentrations occurred at several beaches along the Pacific coast of Mexico, including Piedra del Tlacoyunque (Guerrero), Bahia Chacahua, and El Playon de Mismaloya (Jalisco), but in recent years the Mexican arribadas have been largely restricted to La Escobilla (Oaxaca), although smaller nesting concentrations have been reported from Morro Ayuta (Oaxaca). In Costa Rica, a major nesting aggregation is found at Ostional, on the Nicoya Peninsula, and smaller arribadas occur at Nancite, in the Santa Rosa National Park. Smaller arribadas also occur in Nicaragua at La Flor (Ruiz 1994) and Chacocente and at several localities in Panama. In the Indian Ocean, four arribada sites have been reported in the Indian State of Orissa, the most important being Gahirmatha Beach. In the Atlantic, only small arribadas, numbering at most a few hundred animals per night, have been reported from a single locality (Eilanti, Surinam).

Arribadas may be precipitated by such climatic events as a strong offshore wind (in Surinam) (Pritchard 1969), or by certain phases of the moon and tide (at Nancite, Costa Rica)

(Cornelius 1986), but there is a major element of unpredictability at all arribada sites. This unpredictability, and the apparent ability of gravid females to wait for weeks while holding fully-shelled eggs, may be an important aspect of the survival advantage of arribada-formation, a phenomenon usually interpreted as one that evolved as a predator-saturation device.

Individual olive ridleys may nest one, two or three times per season (Pritchard 1969, Plotkin 1994), typically producing 100-110 eggs on each occasion. The internesting interval is variable, but for most localities it is approximately 14 days for solitary nesters and 28 days for arribada nesters (Pritchard 1969, Kalb and Owens 1994, Plotkin 1994). The genus is also unique in that ridleys of both species commonly, and probably typically, nest each year, without intervening non-breeding seasons as shown by dermochelyids and other cheloniids. The ridleys nesting in an arribada could not be sustained by the productivity of immediately adjacent marine ecosystems, and the species is indeed migratory. Pritchard and Trebbau (1984) have documented the post-nesting dispersal of Surinam ridleys to the waters of all of the Guianas, Trinidad, and eastern Venezuela as far as Isla Margarita. Cornelius and Robinson (1986) illustrated comparable dispersal of olive ridleys from Nancite, Costa Rica, to the waters of all countries from Mexico to Peru, with some open sea recoveries, the farthest being 2,400 km west of Costa Rica. Results of recent investigations (Arenas and Hall 1990, Pitman 1990, Plotkin et al. 1993, Plotkin 1994) support previous contentions (Hendrickson 1980) that olive ridleys reside in oceanic habitats of the eastern Pacific Ocean during the non-reproductive portion of its life cycle. Plotkin (1994) documented a nomadic migratory behavior of female and male olive ridleys from the reproductive aggregation at Nancite, Costa Rica and suggested that this pattern evolved in response to spatio-temporal variations in productivity and available food resources of the eastern Pacific Ocean.

The overall distribution of the olive ridley shows interesting parallels with that of the utterly different leatherback turtle (*Dermochelys coriacea*). Both occupy oceanic habitats (see Morreale et al. 1993 for leatherback migrations) and both nest primarily on Pacific shores of the American tropics and in the Guianas, in moderate numbers in tropical West Africa, and in relatively small numbers elsewhere, both being extremely rare, for example, throughout Australia, or Pacific oceanic islands.

Despite its local abundance, there are surprisingly few data relating to the feeding habits of the olive ridley. However, those reports that do exist (Caldwell et al. 1969, Carr 1961, Márquez et al. 1976, see Mortimer 1982 for review) suggest that the diet in the western Atlantic and eastern Pacific includes crabs, shrimp, rock lobsters, jellyfish, and tunicates. In some parts of the world, it has been reported that the principal food is algae (Carr 1961, Biswas 1982).

Population Size

Because of the continued existence of several large arribadas, it is probable that the olive ridley is, in terms of absolute numbers of adult individuals in existence, the most abundant sea turtle species in the world. Nevertheless, there is evidence of downward trends at several arribada beaches. The various populations are under considerable stress, and the concentration of such a large proportion of the reproductive animals into a few arribadas may be a liability, not only in that such aggregation facilitates industrial-scale exploitation, as it has in Mexico as well as on the feeding grounds in Ecuador, but also because arribadas do not seem to be an efficient method of guaranteeing maximum reproductive efficiency. Indeed, at the relatively undisturbed arribada beach of Nancite, within Santa Rosa National Park, Costa Rica, it has been estimated by Cornelius (1986) that only about 5 % of eggs laid actually produce hatchlings, and D. Robinson (pers. comm.) has suggested that, under natural conditions, it may be solitary nesters, many fewer though they may be, that bear principal responsibility for the reproduction of the species. Contrary to Robinson's hypothesis, Eckrich and Owens (1995) recently found that significantly more first-night solitary nests were predated by mammalian predators than arribada nests at Nancite, thereby suggesting that solitary nests contribute little to the overall reproduction of the species at this beach. It is unknown if mammalian predation adversely impacts solitary nests laid on non-arribada beaches and if so, to what extent this occurs.

The number of ridleys nesting during an arribada is difficult to count, although methodologies to estimate arribada size have been developed that are useful if nesting is well supervised by competent biologists (Cornelius and Robinson 1985, Gates et al. *In press*). On the other hand, estimates by laymen of numbers of turtles in a given arribada are probably so inaccurate as to be useless.

Because nesting in successive years is commonplace for olive ridleys (Pritchard 1969, Cornelius 1986, Plotkin 1994), and may well be the norm for the species, the erratic nesting population trend lines often shown by loggerhead or green turtle populations, that very rarely nest in successive years, are not shown by olive ridley populations. It is thus much easier and more justified to draw conclusions about overall ridley population trends from a few years of comprehensive nest counts than it is for those species with multi-year nesting cycles.

There is a little evidence to suggest that olive ridley arribadas may not be permanent phenomena, but rather may be subject to shifts over the years from one site to another. Thus, the Mexican arribadas at Playon de Mismaloya and Piedra de Tlacoyunque disappeared within a few years of their discovery in the 1960s, even though it was the more southerly nesting population at Escobilla that was subjected to the most intensive industrial slaughter, and that still survives. Moreover, older inhabitants of the village of Ostional, located behind the arribada beach of Ostional, Costa Rica, seem to unite in the opinion that arribadas first occurred at that site in or around 1961 (S. Cornelius pers. comm.).

Published estimates for the sizes of nesting populations in the Western Atlantic, Costa Rica, Mexico, and India are as follows:

Data to document the declining western Atlantic population are clear and unequivocal. Although Pritchard found shells of twenty or more beach-slaughtered olive ridleys on a visit to Shell Beach, Guyana, in 1964, in the course of the last eight seasons (1987 - 1994) intensive season-long patrols have recorded only one or two nestings per year. Nesting was never abundant in French Guiana, and the rumored occurrence of small arribadas at Kourou does not seem to have been based on fact. Data given by Reichart (1993) indicate that 31 nestings were recorded in French Guiana in 1977, 101 in 1978, 33 in 1979, 6 in 1980, no data for 1981 to 1985, 52 in 1986, and 479 in 1987. The key country for ridleys in the west Atlantic region is Surinam, and there the data show an alarming downward trend (Table 1).

At Nancite, Costa Rica, a 1982 estimate (quoted by Mager 1985) suggested that 221,000 ridleys nested there each year. Estimates of the number of female ridleys nesting at Nancite show a decrease in the population size during the 1980's (Valverde et al. 1990). Most recent estimates show a significant decrease: an estimated 37,123 ridleys nested at Nancite in 1990 (Zanella and Mo 1990), 34,189 nested in 1991 (Calvo and Mo 1991), and 31,029 nested in

1992 (Torre and Mo 1992). While a precipitous decline has occurred at Nancite, the number of females nesting at Ostional, approximately 90 km away, appears to be stable and possibly increasing. Mager (1985) provided a 1982 Ostional estimate of between 260,000 and 435,000 nesting ridleys.

Pacific Mexico supported an estimated 1,185,000 adults (including 593,667 females) prior to 1969; and 485,000 adults in 1976 (Cliffon et al. 1982). Large arribadas occurred at El Playon de Mismaloya (Jalisco), Piedra del Tlacoyunque (Guerrero), Bahia Chacahua, La Escobilla (Oaxaca), and El Morro Ayuta (Oaxaca). Several populations collapsed during the 1970s (Tlacoyunque, Chacahua, Mismaloya) (Ross 1982) and in the early 1970s only between 179,000 and 400,000 nesting ridleys arrived at the remaining arribada beaches (Groombridge 1982). J. Woody (pers. comm. to A. Mager) estimated the Mexican population at 153,300 adults in 1981, 164,200 in 1982, and 79,900 in 1983. At La Escobilla, the only remaining site in Mexico where large arribadas still occur, an estimated 202,470 ridleys nested in 1987 (Ruiz and Marin 1988). An estimated 200,000 + female olive ridleys nested at La Escobilla during the September 1995 arribada and it is expected that the total number of turtles nesting during the 1995-1996 season will exceed 500,000 turtles (R. Byles, pers. comm., 1995).

Although nesting in a dispersed fashion is very widespread along the eastern coast of India south to Sri Lanka, and to a lesser degree along the west coast as well, the great majority of olive ridleys nest in four arribadas in Orissa. Gahirmatha, located in the Bhitarkanika Wildlife Sanctuary, Orissa, adjacent to the western Bay of Bengal, is perhaps the most important arribada beach in India. The estimated number of ridleys nesting during arribadas at Gahirmatha since 1984 has fluctuated from as little as no turtles to as many as 805,000 individuals (mean = 398,636 turtles) (Mohanty-Hejmadi and Sahoo 1994).

Listing Factors

In the western Atlantic, nesting of the olive ridley outside Surinam is desultory and, as far as is known, has never been abundant. Accordingly, numerical trends of nesting animals in Surinam, where turtle beaches are closely monitored by STINASU, holds the key to population evaluation in the entire western Atlantic.

The nesting beaches themselves (principally Bigisanti and Eilanti) have been well protected by personnel from STINASU, the Surinam nature protection foundation, for over two decades. Consequently, it seems that some at-sea stress is depleting the population despite good annual production of hatchlings and effective protection of nesting adults. The tag returns of adult turtles that nested in Surinam indicates post-nesting dispersal to feeding grounds extending from eastern Surinam and western French Guiana through Guyana to Eastern Venezuela (Isla Margarita, Trinidad, etc.) (Pritchard and Trebbau 1984). This entire area is heavily trawled for shrimp, and trawlers have been the principal source of returns of tags from this nesting population. Consequently, it appears inescapable that incidental capture in trawls has been the cause of the progressive depletion of this population.

Although comprehensive data to back up this conclusion are not available, Reichart (1989) reported that, in the 1960s, a single ship engaged in periodic experimental trawling exercises along the coast of the Guianas caught 39 olive ridleys in the course of a one year period. A rough estimate indicated that 90-100 shrimp boats operate off the coast of French Guiana, and an ex-crewmember interviewed by Reichart suggested that the typical incidental catch of ridleys was 2-3 animals per day. Even larger numbers of shrimp trawlers operate out of Guyana, and some indication of the magnitude of the problem thus becomes evident. We also know for sure that ridleys have frequently been caught by trawlers operating in waters of eastern Venezuela, as indicated by the returns of tags (Pritchard and Trebbau 1984). TEDs are currently required and in use in this area, and it is hoped that with sufficient enforcement of regulations, high mortality of sea turtles in trawls will no longer occur.

In the eastern Pacific, the situation is different and more complex. Solitary nesting by olive ridleys occurs along an enormous coastline extending from Sonora, Mexico, to Colombia. Few of these beaches are patrolled by conservation crews, and egg collection by local people is intensive throughout the area. In many parts there are as many egg collectors as there are turtles, and no nests escape. Even so, it is relatively rare for nesting turtles to be killed along eastern Pacific shores. On the other hand, at a few points including Escobilla and Morro Ayuta, Mexico; La Flor and Chacocente, Nicaragua; and Nancite and Ostional, Costa Rica, the turtles nest in arribadas, and these sites of intensive nesting are subject to at least some measure of protection, though this varies greatly from one site to another. A critical unknown relates to the intensity of incidental capture of olive ridleys at sea by the recently banned high-seas driftnet fishery, and the current shrimp fishery, and long-line

fisheries in the eastern Pacific Ocean. Trawling is not universal throughout this area, but it is intensive in Mexico, Guatemala, El Salvador, Nicaragua, Costa Rica, and Panama. Estimated turtle mortality from Guatemala to Costa Rica is 60,042 turtles per year, with most of these mortalities presumably being olive ridleys (Arauz In press). TEDs are not currently required, nor in use, but there are tentative agreements pending among many eastern Pacific countries and TEDs may be required in the foreseeable future. The long-line fishery of the eastern Pacific likely also poses a significant threat to olive ridleys. A large fleet of vessels operates from Mexico to Panama in nearshore to offshore waters. Although there are no quantifiable estimates available, this indiscriminate fishery often encounters, entangles, hooks, and kills olive ridleys in the eastern Pacific Ocean (Plotkin, pers. comm.). Olive ridleys are also captured incidentally in the western Pacific long-line fishery which operates near the Hawaiian Islands (P. Plotkin, pers. comm.).

Natural phenomena such as El Niños are probably very significant to the survival status of olive ridleys in the eastern Pacific Ocean. It was recently reported (Hill 1995) that warming trends in the Pacific, caused by 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. Similarly, warming in the eastern tropical Pacific might be partially responsible for the decline in the number of olive ridleys nesting along middle America.

Recent information from several nations along the eastern Pacific coast of the Americas gives insight into the current status of the olive ridley, as follows.

Mexico

In Mexico, after several decades of legal, industrial-scale take of olive ridleys, mainly for export of leather products to Japan, the President of Mexico imposed a complete and indefinite closed season on the utilization of sea turtles within its national jurisdiction (Aridjis 1990). The decree was signed by the Secretary of SEDUE (Urban Development and Ecology) and the Secretary of Fisheries at Chapultepec Castle in the presence of the governors of the important coastal states of Jalisco, Michoacan, Guerrero, and Oaxaca. Commercial and industrial fishermen had until 12 June 1990 to file inventories of existing

stocks of turtle products with the Fisheries Department. These inventoried products may continue to be marketed under strict conditions until stocks are depleted.

An Associated Press article (published in the Milwaukee Journal, Nov. 25 1990) commented on the changes that had occurred in Mexico as a result of the decree. The primary slaughterhouse at San Agustinillo was closed and its roof had fallen in. On the other hand, clandestine egg collection continues, and although the fleets of turtle boats are no longer seen near the nesting beach at Escobilla, there are reports of two clandestine slaughterhouses.

The legal quota for olive ridley capture in 1989 was 23,000, but a total (legal plus illegal) of as many as 70,000 may have been taken. The reported progressive increase in nesting for the last three seasons is thus all the more surprising in view of the reported level of slaughter during that period.

Steiner and Tweeten (1990) provide additional first-hand information on the status of the turtle industry in Mexico in late 1990. They confirmed that the San Agustinillo slaughterhouse had closed, but that retail sales of turtle products continues. One dealer in the Tepito market reported an inventory of 1000 pairs of turtle-skin boots. Although aware of the closure of the fishery and the requirement for filing an inventory of existing stocks, he had not filled out any paperwork as required by the May 28 decree, and felt confident that clandestine take would continue to supply the turtleskin boot industry. Turtle eggs were also seen for sale in the Tepito Market (5000 pesos for three), and rumors were heard of certain families continuing to capture large numbers of turtles, and new boats with dual 75 hp engines were still being acquired for that purpose. Sales of sea turtle products in Mexico have continued despite the 1990 decree. Steiner et al. (1994) recently reported that the number and value of sea turtle products confiscated at the U.S.-Mexico border prior to 1990 have not changed significantly since 1990.

El Salvador

In El Salvador, no arribadas have been identified, but there is widespread low-density ridley nesting. It is estimated that only about 2 % of the nests laid escape predation by humans (C. R. Hasbun, pers. comm.). In addition, it is reported that dead adult olive ridleys often wash ashore on the 321 km of the Salvadorean coast. These are reportedly the victims of shrimp

trawlers, which are required by law to operate at least three miles from the shore, but which in fact trawl within a few yards of shore much of the time.

Nicaragua

A Reuters report, filed September 25 1990, gives some information on the status of ridley nesting in Nicaragua. The arribada beach at Chacocente, where groups of up to about 4000 ridleys come ashore to nest six or eight times per year, is patrolled by police armed with machine guns and dressed in battle fatigues and combat boots. For years, virtually every egg was collected immediately and sold to middlemen on the spot, who would then sell the eggs to local restaurants or export them. However, a new decree bans turtle egg collecting throughout the entire season (July 1 to January 31) at Chacocente and five other Pacific nesting sites. The absolute ban has not been well received by the local people, who in some cases have registered their displeasure by killing turtles caught at sea and stripping them of their oviducal eggs. A new proposal, to allow egg collection during the first day (only) of each arribada, is now under consideration.

Costa Rica

In Costa Rica, some protection of olive ridleys occurs at Nancite by virtue of its remote location and presence in a National Park. The decline in this nesting population is cause for concern and may indicate an overall decline of olive ridleys in the eastern Pacific. The larger nesting colony at Ostional is managed for egg production for human consumption, a scheme that shows great promise and may actually increase the number of hatchlings produced, but has not yet been definitively evaluated. Hatchling predation by black vultures (*Coragyps atratus*) is sufficiently intense as to suggest the need for some kind of control measures.

Ecuador

In Ecuador, there was formerly a massive directed take of olive ridleys caught on the feeding grounds near the Colombian border; nesting of ridleys this far south is desultory. Although there has not been much recent information on the status of the olive ridley harvest in Ecuador, a decree (dated July 31 1990) by the Ecuadorian Subsecretariat for Fishery

Resources, following a recommendation by the Charles Darwin Foundation for the Galapagos islands, affords complete protection to all species of sea turtles in Ecuadorian waters, and prohibits the capture, processing, and internal and overseas trade, in all sea turtle species.

West Africa and Indian Ocean

No current data are available to assess the status of olive ridleys in West Africa.

Large nesting aggregations occur in the northern Indian Ocean and Bay of Bengal. Mohanty-Hejmadi and Sahoo (1994) reported on the size of arribadas at Gahirmatha Beach, the largest nesting aggregation of ridleys in the world, from 1984 to 1994. The estimated number of turtles nesting at Gahirmatha per season has ranged from zero (no mass nesting) to as many as 805,000 ridleys. The most current estimates suggest that this population is relatively stable, but development and enhancement of shrimp trawl fisheries in this region threaten to change its status. In the Bay of Bengal, along the Orissa coast, several thousand olive ridleys may be incidentally captured and killed in shrimp trawl nets annually. Some olive ridleys are also incidentally captured and killed in the gill net fishery which operates near the coast. Over 5,000 dead ridleys stranded along the Orissa coast from December 1993 through April 1994 (Pandav et al. 1994). New jetties and fishing ports constructed adjacent to Gahirmatha (Mohanty-Hejmadi and Sahoo 1994, Eckert and Eckert 1994) undoubtedly will attract more boats to the area and threaten havoc on this population if sea turtle conservation measures are not made mandatory.

Conclusions

The status of the olive ridley in the western Atlantic is unambiguous. Populations are very low and continue to decline, almost certainly as a result of long-standing incidental capture in shrimp trawls. This population is in worse condition than that of the endangered Kemp's ridley (*L. kempii*) and should clearly be listed as ENDANGERED.

Available data are too few to assess the precise survival status of the species in the eastern Atlantic and northern Indian Oceans. However, probable widespread egg collection throughout West Africa and known problems with incidental capture and mortality in shrimp

trawlers and gill nets near the arribada beaches of Orissa, India seem to indicate that the populations there are stressed and should continue to be listed as THREATENED.

In the eastern Pacific, data indicate that some nesting aggregations are in decline, while others appear relatively stable. There currently are no controls at all on incidental capture by shrimp trawl and long-line fisheries throughout this vast region, which appears significant throughout much of the region. We recommend continuation of the status quo listing of the olive ridley in the eastern Pacific as THREATENED.

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Table 1. Annual nest counts and estimated population size for the Suriname olive ridley turtle (*Lepidochelys olivacea*) (Reichart 1993).

<u>Year</u>	<u># Nests</u>	<u># Nesting Females</u>
1967	2675	1440 - 2050
1968	3290	1650 - 2350
1969	1665	830 - 1190
1970	1750	880 - 1250
1971	1595	800 - 1140
1972	1270	640 - 910
1973	890	450 - 640
1974	1080	540 - 770
1975	1070	540 - 760
1976	1160	580 - 830
1977	1030	520 - 740
1978	870	440 - 620
1979	795	400 - 570
1980	1020	510 - 730
1981	1220	610 - 870
1982	1045	520 - 750
1983	1212	610 - 870
1984	940	470 - 670
1985	670	340 - 480
1986	540	270 - 390
1987	659	330 - 470
1988	569	280 - 400
1989	585	290 - 420