

2004 GLOBAL STATUS ASSESSMENT

Green turtle (*Chelonia mydas*)



**Marine Turtle Specialist Group
The World Conservation Union (IUCN)
Species Survival Commission
Red List Programme**

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Class: Reptilia; **Subclass:** Anapsida; **Order:** Testudines; **Family:** Cheloniidae; **Subfamily:** Chelonini

Taxon Name: *Chelonia mydas* (Linnaeus 1758)

Common Names: Green turtle (English); tortue comestible, tortue franche, tortue verte (French); tortuga verde, tortuga blanca (Spanish); tartaruga verde, aruanã (Portugese).

Status: Endangered globally (EN A2bd; IUCN 2001a)

Distribution: Multiple genetic stocks occurring worldwide in tropical and subtropical marine waters.

Range:

Circumglobal, tropical to subtropical seas. Nests in over 80 countries worldwide.

Habitats:

Adults nest on sandy beaches; posthatchlings, small juveniles, and migrating adults occur in oceanic zones; larger juveniles and adults forage in neritic habitats.

Threats:

Primary threats include long-term harvest of eggs and adults at nesting beaches and capture of juveniles and adults at feeding areas. Secondary threats include incidental capture in marine fisheries, habitat loss at nesting and foraging areas, and disease.

Rationale. Analysis of historic and recent published accounts indicate extensive subpopulation declines in all major ocean basins over the last three generations as a result of overexploitation of eggs and adult females at nesting beaches, juveniles and adults in foraging areas, and, to a lesser extent, incidental mortality relating to marine fisheries and degradation of marine and nesting habitats. Subpopulation declines of over 50 % have been identified in the eastern Atlantic Ocean (Bioko Is., Equatorial Guinea), western Atlantic Ocean (Aves Is., Venezuela), Southeast Asia (Suka Made, Indonesia; Terengganu, Malaysia), northern Indian Ocean (Gujarat, India; Hawkesbay and Sandspit, Pakistan; Sharma, Peoples Democratic Republic of Yemen), and western Indian Ocean (Seychelles Republic). Declines greater than 80 % have been shown for subpopulations in the eastern Pacific Ocean (Colola, México), Asia (Berau Islands and Pangumbahan, Indonesia; Sarawak, Malaysia), northeastern Indian Ocean (Thamihla Kyun, Myanmar), and Mediterranean Sea (Turkey). In all cases declines have occurred in less than three generations, suggesting that absolute reductions over the entire 3-generation time spans are much greater.

Information on nesting activity over the last three decades indicates that green turtle subpopulations are currently stable or increasing in Ascension Island, Australia, Comoros Islands, Costa Rica (Tortuguero), Ecuador (Galápagos Islands), Guinea-Bissau (Bijagos Islands), Malaysia (Sabah), México (Yucatan Peninsula), Oman (Ras al Hadd), Saudi Arabia (Karan Island), Suriname, and the United States. However, the statuses of these subpopulations relative to populations three generations ago are unknown, and several face substantial threats of mortality through poaching, fisheries impacts, habitat loss, and disease (Table 6).

Despite increasing conservation attention to green turtles, intentional harvest continues worldwide. Egg collection is ongoing at nesting beaches in the eastern Atlantic Ocean (Fretey

1998; 2001), western Atlantic Ocean (van Tienen et al. 2000), Caribbean (Mangel et al. 2001), southern central Pacific Ocean (Eckert 1993), eastern Pacific Ocean (Alvarado et al. 2001), and Southeast Asia (Cruz 2002, Dermawan 2002, Liew 2002, Sharma 2002). Nesting females continue to be killed in the Caribbean Sea (Fleming 2001, Mangel et al. 2001), eastern Atlantic Ocean (Fretey 2001), Southeast Asia (Cruz 2002), and Indian Ocean (Humphrey and Salm 1996). Of perhaps greatest current threat to the stability of existing green turtle stocks is the intentional capture of juveniles and adults at neritic foraging habitats (National Marine Fisheries Service and U. S. Fish and Wildlife Service 1991; 1998a; 1998b). High levels of take are present in the eastern Atlantic Ocean (Formia 1999), Caribbean Sea (Lagueux 1998), Indian Ocean (Humphrey and Salm 1996, Andrew Cooke pers. comm. to J. Mortimer), Mediterranean Sea (Kasperek et al. 2001), central Pacific Ocean (Eckert 1993), eastern Pacific Ocean (Seminoff 2000, Nichols 2001, Gardner and Nichols 2001), and Southeast Asia (Pilcher 1999, Limpus et al. 2002).

Because of slow maturation rates for green turtles, the effects of egg and juvenile mortality have yet to manifest fully at nesting beaches. Although large numbers of females continue to nest in many areas, egg harvests decrease the recruitment and overall abundance of juveniles, thus hindering this age-group's ability to replace aging adults. Declining population trends are exacerbated when harvest is more intense or longer term (Chaloupka 2000), and when nesting females are also exploited.

The genetic substructure of the green turtle regional subpopulations shows distinctive mitochondrial DNA properties for each nesting rookery (Bowen et al. 1992). Mitochondrial DNA data suggest that the global matriarchal phylogeny of green turtles has been shaped by ocean basin separations (Bowen et al. 1992, Encalada et al. 1996) and by natal homing behavior (Meylan et al. 1990). The fact that sea turtles exhibit fidelity to their natal beaches suggests that if subpopulations become extirpated they may not be replenished by the recruitment of turtles from other nesting rookeries over ecological time frames. Moreover, because each nesting subpopulation is genetically discrete, the loss of even one rookery represents a decline in genetic diversity and resilience of the species (Bowen 1995).

The loss of ecological function due to depletion of these large, long-lived animals may have serious implications for the maintenance of both marine and terrestrial ecosystems. As large herbivores, green turtles impact seagrass productivity and abundance (Bjorndal 1980, Ziemann et al. 1984) and continue to represent an essential trophic pathway over expansive coastal marine habitats (Thayer et al. 1982; 1984, Valentine and Heck 1999). Through egg deposition on beaches, sea turtles act as biological transporters of nutrients and energy from marine to terrestrial ecosystems (Bouchard and Bjorndal 2000). Thus, as green turtle stocks are depleted we can expect a corresponding breakdown in the health of coastal marine and terrestrial systems (Jackson 1997, Jackson et al. 2001).

The green turtle has been a species of global concern for decades, and was previously listed by IUCN as *Endangered* (Groombridge 1982, Baillie and Groombridge 1996, Hilton-Taylor 2000). The majority of the most important nesting populations of green turtles have declined in the 20th century at substantial rates. Although a few large subpopulations remain, they are vulnerable to exploitation, incidental capture in marine fisheries, habitat loss, and disease.

Analyses of subpopulation changes at 32 Index Sites distributed globally (Fig. 1, Table 1) show a 48% to 67% decline in the number of mature females nesting annually over the last 3-generations. These estimates are, however, based on a conservative approach; actual declines may exceed 70 %. This rate of decline, coupled with impending threats (Table 6), justifies

Endangered status for green turtles under the 2001 Red List Criteria. Further, during the present assessment process it became clear that there are different regional patterns in green turtle subpopulation growth trajectories.

Range & Population. The green turtle has a circumglobal distribution, occurring throughout tropical and, to a lesser extent, subtropical waters (Atlantic Ocean – eastern central, northeast, northwest, southeast, southwest, western central; Indian Ocean – eastern, western; Mediterranean Sea; Pacific Ocean – eastern central, northwest, southwest, western central). Green turtles are highly migratory and they undertake complex movements and migrations through geographically disparate habitats. Nesting occurs in more than 80 countries worldwide (Hirth 1997). Their movements within the marine environment are less understood but it is believed that green turtles inhabit coastal waters of over 140 countries (Groombridge and Luxmoore 1989).

The primary nesting rookeries (i.e., sites with ≥ 500 nesting females per year) are located at Ascension Island (Mortimer and Carr 1987), Australia (eastern, Limpus 1980; western, Prince 1983), Brazil (Trindade Island, Moreira et al. 1995), Comoros Islands (Frazier 1985), Costa Rica (Tortuguero, Carr et al. 1982, Bjorndal et al. 1999), Ecuador (Galápagos Archipelago, Green 1983), Equatorial Guinea (Bioko Island, Tomas et al. 1999), Guinea-Bissau (Bijagos Archipelago, Barbosa et al. 1998), Isles Eparces (Tromelin Island, LeGall et al. 1986; Europa Island, Legall et al. 1986), Indonesia (Schulz 1987), Malaysia (de Silva 1982), Myanmar (Kar and Bhaskar 1982), Oman (Ross and Barwani 1982), Philippines (de Silva 1982), Saudi Arabia (Miller 1989), Seychelles Islands (Mortimer 1984), Suriname (Schulz 1982), and United States (Florida, Ehrhart and Witherington 1992; Hawaii, Balazs 1980).

Lesser nesting areas are located in Angola (Carr and Carr 1991), Bangladesh (Khan 1982), Bikar Atoll (Fosberg 1990), Brazil (Atoll da Rocas, Bellini et al. 1996), Chagos Archipelago (Mortimer and Day 1999), China (Groombridge and Luxmoore 1989), Costa Rica (Pacific Coast, Cornelius 1982), Cuba (Nodarse et al. 2000), Cyprus (Kasperek et al. 2001), Democratic Republic of Yemen (Hirth and Carr 1970), Dominican Republic (Ottenwalder 1981), d'Entrecasteaux Reef (Pritchard 1994), French Guiana (Fretey 1984), Ghana (Fretey 2001), Guyana (Pritchard 1969), India (Kar and Bhaskar 1982), Iran (Tuck 1977), Japan (Suganuma 1985), Kenya (Wamukoya et al. 1996), Madagascar (Rakotoniria and Cooke 1994), Maldives Islands (Frazier 1990), Mayotte Archipelago (Fretey and Fourmy 1996), México (Yucatan Peninsula, Zurita et al. 1994; Michoacán, Alvarado and Figueroa 1990; Revillagigedo Islands, Brattstrom 1982, Awbrey et al. 1984), Micronesia (Wetherall et al. 1993), Pakistan (Kabiraji and Firdous 1984), Palmerston Atoll (Powell 1957), Papua New Guinea (Salm 1984), Primieras Islands (Hughes 1974), Sao Tome é Principe (Brongersma 1982), Sierra Leone (Fretey and Malaussena 1991), Solomon Islands (Vaughan 1981), Somalia (Goodwin 1971), Sri Lanka (Dattatri and Samarajiva 1983), Taiwan (Chen and Cheng 1996), Tanzania (Howell and Mbindo 1996), Thailand (Groombridge and Luxmoore 1989), Turkey (Kasperek et al. 2001), Scilly Atoll (Lebeau 1985), Venezuela (Medina and Solé as cited in Ogren 1989), and Vietnam (Hien 2002). Sporadic nesting occurs in at least 30 additional countries (Groombridge and Luxmoore 1989).

How has human influence shaped today's distributions?

The present distribution of the breeding sites has been largely affected by historical patterns of human exploitation. The only substantial breeding colonies left today are those that have not been permanently inhabited by humans or have not been heavily exploited until recently (Groombridge and Luxmoore 1989). This demographic trend is corroborated by the fact that

several islands which formerly held large breeding colonies are known to have lost them once becoming inhabited by humans (e.g. Bermuda, King 1982; Mauritius, Hughes 1982; Reunion, Bertrand et al. 1986; Cape Verde Islands, Parsons 1962). In addition, the Cayman Island rookery, formerly one of the largest green turtle rookeries in the world, was nearly if not totally extirpated after human colonization and the onset of an organized turtle fishery at these islands (Lewis 1940, Parsons 1962). Although green turtles continue to nest at extremely low levels at these islands (Aiken et al. 2001), it is unknown whether they are a relict nesting subpopulation or the result of re-colonization by turtles from adjacent nesting rookeries in the western Atlantic or head started turtles from the Cayman Turtle Farm (Wood and Wood 1993). Nonetheless, these examples illustrate the broad-reaching effects of human exploitation and underscore the need for effective, long-term conservation to prevent green turtles from declining further.

Taxonomic structure. The genetic substructure of the green turtle regional subpopulations shows distinctive mitochondrial DNA properties for each nesting rookery (Bowen et al. 1992). Mitochondrial DNA data suggest that the global matriarchal phylogeny of green turtles has been shaped by ocean basin separations (Bowen et al. 1992, Encalada et al. 1996) and by natal homing behavior (Meylan et al. 1990). Within the eastern Pacific Ocean, specific or subspecific status has been applied to green turtles (also known as black turtles; *C. (=mydas) agassizii*) ranging from Baja California south to Peru and west to the Revillagigedos Islands and Galápagos Archipelago (Márquez 1990, Pritchard 1997); however, genetic analyses do not support such taxonomic distinctiveness (Bowen et al. 1992, Karl et al. 1992).

Generation Length. Generation length is based on the age to maturity plus one half the reproductive longevity (Pianka 1974). Although there appears to be considerable variation in generation length among sea turtle species, it is apparent that all are relatively slow maturing and long-lived (Chaloupka and Musick 1997). Green turtles exhibit particularly slow growth rates, and age to maturity for the species appears to be the longest of any sea turtle (Hirth 1997). As a result, this assessment uses the most appropriate age-at-maturity estimates for each index site. At Index Sites for which there are local age-to-maturity data, those data are used to establish generation length. When data are lacking, as they are for a majority of subpopulations, information from the closest subpopulation for which data are available are used to generate age-at-maturity estimates (Table 2).

Estimates of reproductive longevity range from 17 y to 23 y (Carr et al. 1978, Fitzsimmons et al. 1995). Data from the apparently pristine green turtle stock at Heron Island in Australia's southern Great Barrier Reef show a mean reproductive life of 19 y (Chaloupka et al. 2004). Because Heron Island is the only undisturbed stock for which reproductive longevity data are available (M. Chaloupka, pers. comm.), this datum is used for all Index Sites (Table 3). Thus, based on the range of ages-at-sexual-maturity (26 yrs to 40 yrs) and reproductive longevity from the undisturbed Australian stock (19 yr), the generation lengths used for this assessment range from 35.5 yrs to 49.5 yrs (Table 3).

Habitats. Like most sea turtles, green turtles are highly migratory and use a wide range of broadly separated localities and habitats during their lifetimes (for review see Hirth 1997). Upon leaving the nesting beach, it has been hypothesized that hatchlings begin an oceanic phase (Carr 1987), perhaps floating passively in major current systems (gyres) that serve as open-ocean developmental grounds (Carr and Meylan 1980, Witham 1991). After a number of years in the

oceanic zone, these turtles recruit to neritic developmental areas rich in seagrass and/or marine algae where they forage and grow until maturity (Musick and Limpus 1997). Upon attaining sexual maturity green turtles commence breeding migrations between foraging grounds and nesting areas that are undertaken every few years (Hirth 1997). Migrations are carried out by both males and females and may traverse oceanic zones, often spanning thousands of kilometers (Carr 1986, Mortimer and Portier 1989). During non-breeding periods adults reside at coastal neritic feeding areas that sometimes coincide with juvenile developmental habitats (e.g., Limpus et al. 1994, Seminoff et al. 2003).

Threats. Green turtles, like other sea turtle species, are particularly susceptible to population declines because of their vulnerability to anthropogenic impacts during all life-stages: from eggs to adults. These impacts are both intentional, such as the harvest of eggs and adults, and accidental, as exemplified by drowning in fishnets. In addition, increased pollution, degradation and loss of coastal and marine habitat, and disease have threatened the stability of ecosystems within which green turtles live (see Table 7).

Intentional Harvests

One of the most detrimental human threats to green turtles is the intentional harvest of eggs from nesting beaches. By taking eggs from nesting beaches, humans have extirpated populations from the bottom up (Mortimer 1995). As each nesting season passes and populations continue to suffer from egg harvest, they will progressively lose the juvenile cohorts that would have recruited from the post-hatchling stock. Present nesting populations may appear hardy, but without recruitment into the juvenile population and a well-balanced distribution of turtles among all cohorts, populations are more vulnerable to decline (Crouse et al. 1987, Frazer 1992). Further, when declines come, they will be fast, thorough, and long-lasting. Directed take of eggs is an ongoing problem in: Comoros Is. (Mohadji et al. 1996), Costa Rica (Tortuguero, Mangel et al. 2001), Guinea (Fretey 2001), Equatorial Guinea (Fretey 2001), Guinea-Bissau (Barbosa et al. 1998), India (Andaman and Nicobar Islands, Andrews 2000), Indonesia (H. Hutabarat pers. comm.), Ivory Coast (Fretey 1998), Malaysia (Terengganu, Limpus 1995), Maldives (H. Zahir pers. comm.), México (Alvarado-Díaz et al. 2001), Panama (Evans and Vargas 1998), Philippines (Cruz 2002), Sao Tome é Principe (Fretey 2001), Saudi Arabia (Karan Island, Pilcher 2000, Al-Merghani et al. 2000), Senegal (Fretey 2001), Sri Lanka (T. Kapurusinghe pers. comm.), Thailand (Limpus 1995), Vietnam (P. Thuoc pers. comm.), and the Pacific Islands of American Samoa, Guam, Palau, Commonwealth of the Northern Mariana Islands, Federated states of Micronesia, Republic of Marshall Islands, and the Unincorporated Islands Iwake, Johnston, Kingman, Palmyra, Jarvis, Howland, Baker, and Midway (Eckert 1993). The above list is by no means comprehensive but it does, however, illustrate the widespread nature of this problem.

In addition to the collection of eggs from nesting beaches, the killing of nesting females continues to threaten the stability of green turtle subpopulations. As mentioned previously, this affects subpopulations both by depleting the current subpopulation and through reducing the subpopulation's egg producing potential. Ongoing harvest of nesting adults has been documented at Bioko Island (J. Tomas pers. comm.), Costa Rica (Mangel et al. 2001), Guinea Bissau (Fortes et al. 1998), India (Andaman and Nicobar Islands, Andrews 2000), Japan (Y. Matsuzawa pers. comm.), México (Michoacán, Alvarado-Díaz et al. 2001), western Australia (R. Prince pers. comm.), Seychelles (Mortimer et al. 1996), and Yemen (Saad 1999). Although there

are likely more countries at which such harvests continue, it is apparent, based on the above list, that harvest of nesting females remains a problem in many areas throughout the world.

Mortality of turtles in foraging habitats continues to be problematic for recovery efforts worldwide. Although subpopulations may be protected at nesting beaches, their large-scale in-water movements often traverse arbitrary national boundaries and take them to areas where protection is absent. A partial list of the countries that experience ongoing intentional capture of green turtles includes: Australia (Prince 1998), Bahamas (Fleming 2001), British Virgin Islands (Fleming 2001), Cameroon (Fretey 1998), Cayman Islands (Fleming 2001), Comoros Islands (Mohadji et al. 1996), Costa Rica (Tortuguero, Mangel et al. 2001), Cuba (Fleming 2001), Egypt (Nada 2001), Equatorial Guinea (Formia 1999, Tomas et al. 1999), Gabon (Fretey 2001), Ghana (Fretey 2001), Guinea Bissau (Fretey 1998; 2001), India (Andaman and Nicobar Islands, H. Andrews pers. comm.), Indonesia (C. Hitipeuw pers. comm., Limpus et al. 2002), Ivory Coast (Fretey 1998), Liberia (Siakor and Greaves 2001), Madagascar (Rakotonirina and Cooke 1994, Mbindo 1996, A. Cooke pers. comm. to J. Mortimer), Mayotte Archipelago (Fretey and Fourmy 1996), México (Seminoff 2000, Nichols 2001, Gardner and Nichols 2001), New Caledonia (Limpus et al. 2002), Nicaragua (Lagueux 1998), Pakistan (Asrar 1999), southern and eastern Papua New Guinea (Limpus et al. 2002), Sao Tome é Principe (Fretey 1998), Seychelles (Mortimer et al. 1996), Sierra Leone (Fretey 1998), Solomon Islands (Broderick 1998), Togo (Fretey 1998), Turks and Caicos (Fleming 2001), Vanuatu (Limpus et al. 2002), and Vietnam (P. Thuoc pers. comm.). Despite substantial declines in green turtle subpopulation size, harvest remains legal in several of these countries (Humphrey and Salm 1996, Fleming 2001, Fretey 2001).

Incidental Impacts

In addition to the intentional exploitation of green turtles there are increasing incidental threats in the nesting and marine environment that affect green turtles. Structural impacts to nesting habitat include the construction of buildings, beach armoring and re-nourishment, and/or sand extraction (Lutcavage et al. 1997). These factors may directly, through loss of beach habitat, or indirectly, through changing thermal profiles and increasing erosion, serve to decrease the amount of nesting area available to nesting females, and may evoke a change in the natural behaviors of adults and hatchlings (Ackerman 1997). In addition, coastal development is usually accompanied with artificial lighting. The presence of lights on or adjacent to nesting beaches alters the behavior of nesting adults (Witherington 1992) and is often fatal to emerging hatchlings as they are attracted to light sources and drawn away from the water (Witherington and Bjorndal 1990). In many countries, coastal development and artificial lighting are responsible for substantial hatchling mortality. Although legislation controlling these impacts does exist (Lutcavage et al. 1997), a majority of countries do not have regulations in place.

As the human population expands, so do impacts to the coastal zones of both developing and modernized countries. The problems associated with development in these zones will progressively become a greater challenge for conservation efforts, particularly in the developing world where wildlife conservation is often secondary to other national needs. This is underscored by the fact that over the next 40 years the human population is expected to grow by more than 3 billion people (about 50%; United Nations Educational, Scientific, and Educational Organization [UNESCO] 2001). By the year 2025, UNESCO (2001) forecasts that population growth and migration will result in a situation in which 75% of the world human population will

live within 60 km of the sea. Such a migration undoubtedly will change a coastal landscape that, in many areas, is already suffering from human impacts.

Incidental threats do not stop at the nesting beach. Once hatchlings and adults enter the marine environment they are subjected to a myriad of human-related impacts. Although not a direct impact, increased effluent and contamination from coastal development diminishes the health of coastal marine ecosystems and may, in turn, adversely affect green turtles. Sea turtles also suffer directly from incidental interactions with commercial and artisanal marine fisheries. These fisheries practices include drift netting, long-lining, trawling, and dynamite fishing and their adverse impacts on sea turtles have been documented in marine environments throughout the world (e.g., Arauz et al. 1998, Kasparek et al. 2001). Of the world's 17 major fisheries zones, nine are considered depleted and an additional four are in early stages of collapse (Safina 1995). Unfortunately, rather than elicit a closure of fisheries, declines in catch rate are often greeted with new fisheries and expanding fleets (DiSilvestro 1995). Without effective management practices, such expansion likely will result in increased mortality of all sea turtle species.

Disease

Diseases threaten a larger number of existing subpopulations. Certainly the most deleterious of pathogens is Fibropapillomatosis (Herbst 1994). This often-fatal disease has been found in green turtle subpopulations of Australia (eastern, Limpus and Miller 1990; western, Raidal and Prince 1996), Bahamas (K. Bjorndal pers. comm.), Barbados (Gameche and Horrocks 1992), Brazil (Matushima et al. 2000), British Virgin Islands (Overing 1996), Cameroon (Fretey 2001), Cayman Islands (Wood and Wood 1994), Costa Rica (Tortuguero, Mangel et al. 2001), Cuba (Moncada and Prieto 2000), Equatorial Guinea (A. Formia pers. comm.), Federated States of Micronesia (Kolinski 1994), Indonesia (Adnyana et al. 1997), Japan (Y. Matsuzawa pers. comm.), Kenya (R. Zangre pers. comm.), México (Yucatan Peninsula, K. Lopez pers. comm.), Nicaragua (Lagueux et al. 1998), Philippines (Nalo-Ochona 2000), Senegal (Fretey 2001), Seychelles (J. Mortimer pers. comm.), United States (California, MacDonald and Dutton 1990; Florida, Ehrhart 1991; Hawaii, Balazs et al. 1992), U. S. Virgin Islands (Eliazar et al. 2000), and Venezuela (Solé and Azara 1998, Guada and Solé 2000). Epidemiological studies indicate rising incidence of this disease (George 1997), thus the above list will likely grow in the future.

Although Fibropapillomatosis can be considered a natural disease, there is speculation that the prevalence of this disease has reached epidemic proportions due immuno-suppression in green turtles brought about by human-related habitat degradation (George 1997). Clearly, additional studies are necessary to elucidate the causes of this disease, but the fact that human activity has been at least partially implicated in this epidemic suggests that the widespread incidence of Fibropapillomatosis should be taken into consideration when establishing the IUCN Red List status of green turtles.

Conservation measures: Green turtles have been afforded legislative protection under a number of treaties and laws (e.g., Navid 1982, Humphrey and Salm 1996, Fleming 2001, Fretey 2001). Among the more globally relevant designations are those of *Endangered* by the World Conservation Union (IUCN; Baillie and Groombridge 1996, Hilton-Taylor 2000); Annex II of the SPAW Protocol to the Cartagena Convention (a protocol concerning specially protected areas and wildlife); Appendix I of CITES (Convention on International Trade in Endangered Species); and Appendices I and II of the Convention on Migratory Species (CMS). A partial list of the

International Instruments that benefit green turtles includes the Inter-American Convention for the Protection and Conservation of Sea Turtles, the Memorandum of Understanding on the Conservation and Management of Marine Turtles and their Habitats of the Indian Ocean and South-East Asia (IOSEA), the Memorandum of Understanding on ASEAN Sea Turtle Conservation and Protection, the Memorandum of Agreement on the Turtle Islands Heritage Protected Area (TIHPA), and the Memorandum of Understanding Concerning Conservation Measures for Marine Turtles of the Atlantic Coast of Africa.

As a result of these designations and agreements, many of the intentional impacts directed at sea turtles have been lessened: harvest of eggs and adults has been slowed at several nesting areas through nesting beach conservation efforts and an increasing number of community-based initiatives are in place to slow the take of turtles in foraging areas. In regard to incidental take, the implementation of Turtle Excluder Devices has proved to be beneficial in some areas, primarily in the United States and South and Central America (National Research Council 1990). However, despite these advances, human impacts continue throughout the world. The lack of effective monitoring in pelagic and near-shore fisheries operations still allows substantial direct and indirect mortality, and the uncontrolled development of coastal and marine habitats threatens to destroy the supporting ecosystems of long-lived green turtles.

Future actions that are required.

The recovery of green turtles throughout the world will require maximized protection in both nesting and marine environments. Full protection of the remaining nesting beaches is necessary to eliminate poaching of nesting females and eggs, increase egg and hatchling survivorship, and avoid degradation of critical nesting habitat. Because green turtles spend greater than 99 % of their lives in the sea, addressing in-water impacts should also be of high priority (Frazer 1992). As Congdon et al. (1993) discussed with long-lived species, the traits that make green turtles so vulnerable to reduced survival rates also make them very slow to recover once depleted, leaving them vulnerable to other threats even if the impact that initially caused their depletion is addressed. Nest protection efforts may not be sufficient to stop the decline of already threatened subpopulations without the concurrent reduction of human-induced mortality of juveniles and adults in the marine environment (Crouse et al. 1987). Moreover, although hatcheries, head-starting, and captive breeding programs have been used in efforts to increase subpopulations, they remain unproven techniques that merely addresses symptoms rather than actual subpopulation threats. The adoption of such techniques should therefore not be chosen in place of, but rather in coordination with, conservation efforts that directly target the ultimate causes of subpopulation declines (i.e. legal and illegal take, fisheries impacts, and habitat degradation).

The extended longevity and delayed maturity of green turtles dictate that conservation efforts must be long-term in scope (Crowder et al. 1994). Because migratory routes of green turtles commonly cross territorial waters of many nations or occur in the high seas, these practices should involve international collaboration whenever possible.

Recovery efforts will benefit from greater focus on habitat protection and restoration and better enforcement of existing legislation. Coastal seagrass beds and marine algae pastures should be protected. Existing algae harvest practices must be assessed to ensure that practices are sustainable and do not directly impact foraging turtles, particularly the earlier life-stages. Water quality standards should be established and enforced through coastal monitoring efforts. With respect to the distribution of people on the planet, adequate strategies should be established to encourage and legislate ecologically friendly development in coastal zones so as to minimize

the effects of increasing populations and prevent pollution of the marine environment and water resources (UNESCO 2001).

As conservation measures are implemented it is recommended that long-term monitoring programs be established. These may include efforts to track subpopulations at nesting beaches or in foraging habitats. Better monitoring of understudied areas is essential, and research protocols should be standardized so that comparisons can be made within and between sites and the results of monitoring programs must be made available in a timely manner to enable prompt conservation actions (see Eckert et al. 1999).

In the near future, stronger efforts must be put forth to control and reduce intentional take and incidental mortality in marine fisheries. Controlling illegal capture may require increased vigilance at important feeding areas and better monitoring of highways and other human movement corridors used to transport turtle contraband. In the areas that currently experience heavy exploitation, recovery efforts will benefit from the implementation of community-based conservation initiatives. When communities are involved that have a long history of turtle use, conservation efforts should include capacity building and education programs, and provide economic alternatives that are carefully planned and implemented. Whenever possible, local community members should be included early in the planning and decision-making process.

In regard to legal take, careful consideration should be given to cultures that incorporate traditional use into their customs. Efforts should be made to establish and maintain levels of traditional harvest that are sustainable over the long term in these cases. Where, through growth of the coastal population, traditional harvest has become unsustainable, efforts must be made jointly by local scholars, elders and clergy to identify alternate practices. This must be done in a way that balances the cultural integrity of indigenous practices with responsible management of endangered green turtle stocks. Wildlife managers should pursue the best possible understanding of subpopulation sizes and trends to establish what level of take is 'sustainable'. With ongoing traditional practices, adherence to harvest limits may be ensured through periodic monitoring.

Efforts to address incidental capture must be equally broad-based and far-reaching. Such efforts may entail restrictions on, or the elimination of, some fisheries, use of bycatch reduction technologies wherever available, increased frequency of observers onboard fishing vessels, and greater vigilance for vessel adherence to fisheries zones. New fisheries should not be initiated, and current fisheries should not be allowed to expand, until they are carefully analyzed for both target and not target species (Crouse 2000). Moreover, mitigation measures must be built into fishery management plans from the outset.

Assessment Procedure: In accord with the IUCN criterion that Red List Assessments focus on the number of *mature* individuals (IUCN 2001a), this assessment measures changes in the annual number of nesting females. Because reliable data are not available for all subpopulations, the present report focuses on 32 Index Sites (Figure 1, Table 1). These Index Sites include all of the known major nesting areas as well as many of the lesser nesting areas for which quantitative data are available. Despite considerable overlap at some foraging areas, each is presumed to be genetically distinct (Bowen et al. 1992, Bowen 1995) except for the Turtle Islands of Malaysia (Sabah) and Philippines (Moritz et al. 1991). These two Index Sites are, however, treated independently because of the different management practices exercised by the two governments and the resultant differences in subpopulation trends. Selection of the 32 Index Sites was based on two primary assumptions: (1) they represent the overall regional subpopulation trends and (2) the number of individuals among Index Sites in each region is proportional to the actual

population size in that region. Any regional inconsistencies in this proportion may result in a biased global population estimate.

It should be noted that a major caveat of using the number of nesting females to assess population trends is that this data type provides information for the proportion of the adult females that nest in any given year, not the total adult female population. However, when monitored over many years, this index can be reliable for assessing long-term population trends (Meylan 1982, Limpus 1996). In the case of green turtles, which display high inter-annual variability in magnitude of nesting (Limpus and Nichols 1987, Broderick et al. 2001a), using short-term or single-season data sets could misrepresent the actual mean number of nesters over a longer timeframe. To alleviate this potential source of error, we used multiple-year data sets whenever available. However, when single-season datasets represented the only quantitative information for a given time period, these data were used as long as they were in accord with qualitative information from other references.

Because data on annual number of nesting females are not always available, we also used data on number of nests per season, annual hatchling production, annual egg production and annual egg harvest. When these proxies were used, we converted units to number of nesting females based on a constant figure of 100 eggs/nest and three nests/season/female, unless otherwise noted. These conversions were based on the assumptions that (1) the mean number of eggs/nest and nests/female/season differ insignificantly through time, and (2) efforts to monitor nesting female activity and egg production are consistent through time. When using egg harvest data, we also assumed that harvest effort was consistent during all years for which data are available and 100% of the eggs was harvested in any given year. We believe these assumptions are accurate, but their absolute validation is very difficult. Qualitative information does, however, suggest that they are reasonable assumptions. For example, in the case of historic egg harvest, the same group of people usually harvested the eggs at a particular nesting beach each year, and they typically took every egg they could find (e.g. Parsons 1962, Pelzer 1972).

In the present assessment, population abundance estimates are based on raw data, linear extrapolation functions, and exponential extrapolation functions. In most subpopulations, more than one trajectory was exhibited over the 3-generation interval; changes in subpopulation size are thus often based on a combination of raw data and extrapolations. If no change is believed to have occurred outside the time interval for which published abundance data are available, the raw data were used to determine the change in population size. However, when it is believed that change in subpopulation abundance occurred outside the interval for which raw data were available, extrapolations we performed to determine the overall change. Linear extrapolations were used when it was believed that the same amount of change occurred each year, irrespective of total subpopulation size. Exponential extrapolations were used when it was believed that change was proportional to the subpopulation size. In cases where there is a lack of information on the specific rate of change, both linear and exponential extrapolations were used to derive population estimates. However, if extrapolations resulted in obviously false estimates, their results were discarded (see Table 5).

Uncertainties in assessment process: As with any assessment based on historic data or small datasets, there is a great deal of uncertainty relating to the final results of this report. The sources of uncertainty are rooted in both the procedure itself as well as in the stochastic nature green of turtle biology. Both sources of uncertainty are ultimately related to a lack of information, which can be a common issue when dealing with an animal as long-lived as a green turtle.

First and foremost is the uncertainty related to the assumptions invoked for this assessment. For example, if, contrary to our assumption, efforts to monitor nesting female activity and egg production were not consistent through time, then our results may be biased. Similarly, our estimates may be inaccurate if harvest effort or the relative amount of eggs harvested was not consistent through time. Due to a lack of information, it is possible that we did not choose the best extrapolation procedure for all populations. Therefore, the extrapolations in this assessment may also be a source of error. This problem is exacerbated when extrapolations were made over long time intervals or when they were based on short-term data sets.

Uncertainty may also be tied to green turtle biology. In particular, the substantial variability in the proportion of a population that nests in any given year may result in inaccurate comparisons between past and present data sets. For example, if the proportion of a subpopulation's adult female cohort nesting each year oscillates over decadal or longer time frames, then it is conceivable that our estimates of annual change in nesting numbers does not correspond to actual changes in the entire subpopulation. Moreover, if our conversion values for eggs/nest and nests/female/season are not accurate for the specific subpopulation being addressed, inaccuracies may result. Lastly, with respect to the migratory behavior of green turtles, it is expected that each of the Index Sites included in this assessment represent a distinct subpopulation. Indeed, current genetic data support this claim, however, in the absence of complete data for all rookeries, it is possible that turtles moving back and forth between nesting areas in close proximity could have gone undetected. It is thus conceivable that a female could be counted twice. This would, of course, only be a problem when subpopulation size is based on an actual count of individual turtles visiting the beach. Although unlikely, it amounts to an additional source of uncertainty in this assessment.

Population trends. Based on the actual and extrapolated changes in subpopulation size at the 32 Index Sites, it is apparent that the mean annual number of nesting females has declined by 48% to 67% over the last three generations (Table 5). In addition, it is apparent that the degree of population change is not consistent among all Index Sites or among all regions (Tables 5, 6). Because many of the threats that have led to these declines are not reversible and have not yet ceased, it is evident that green turtles face a measurable risk of extinction. Based on this assessment, it is apparent that green turtles qualify for *Endangered* status under Criteria A2bd.

In determining the 3-generation declines for green turtles, the present assessment was conservative in its approach to dealing with uncertainty. The conservative nature of these calculations is evidenced by that facts that:

1. Although it is likely that impacts to subpopulations started long before the earliest documented accounts, many are assumed to have been stable until the first estimate of abundance (i.e., the baselines are from relatively recent times),
2. despite the documented presence of substantial impacts at their respective foraging areas and evidence of decreasing survivorship values for in-water stocks (e.g., Sideek and Baldwin 1996, Limpus et al. 2002, K. Campbell pers. comm.), several Index rookeries (e.g., Oman, eastern Australia, Costa Rica) are classified as *stable* or *increased* based on number of nesting females, and
3. there are a number of formerly large rookeries that are known to have declined in recent years but for which no quantitative data are available that would enable them to be included as Index Sites (e.g., Fiji, Gulf of Carpentaria – Australia, Guyana, Kenya, Somalia,; Parsons 1962)

Recent Documented Declines: A Regional Perspective

Table 6 summarizes the subpopulation trends among 11 regions (based on published *Past* and *Present* estimates, Table 4): (1) eastern Pacific Ocean, (2) central Pacific Ocean, (3) western Pacific Ocean, (4) Southeast Asian seas, (5) eastern Indian Ocean, (6) northern Indian Ocean, (7) western Indian Ocean, (8) Mediterranean Sea, (9) eastern Atlantic Ocean, (10) central Atlantic Ocean, and (11) western Atlantic Ocean and Caribbean Sea. Based on this regional approach it is apparent that green turtle subpopulations exhibit varying overall trends in different parts of the world. For example, green turtle subpopulations in the western Pacific Ocean (Australia), western Atlantic Ocean and central Pacific Ocean are exhibiting encouraging trends: both subpopulations in eastern Australia have increased; all but one nesting subpopulation (Venezuela) in the western Atlantic Ocean are stable or have increased in recent years; and the single rookery examined in the central Pacific Ocean (Hawaii) has increased. In contrast, subpopulations in the Southeast Asia seas, northern and eastern Indian Ocean, eastern Pacific Ocean, and Mediterranean Sea are doing relatively poorly. Among the six rookeries in the Southeast Asian seas, all but one (Sabah, Malaysia) are depleted, and in the northern and eastern Indian Ocean all but two (Saudi Arabia, Oman) have declined. Among rookeries in the eastern Pacific Ocean and Mediterranean Sea, declining trajectories are present at all but one (Galápagos Islands).

Differences in population trajectories among the Index Sites are likely due to variation in both the intensity of historical exploitation and the duration and quality of conservation efforts. In respect to exploitation, patterns of human occupation and the cultural significance of sea turtles have dictated the duration and intensity of green turtle harvests. Rookeries in areas that have had less human presence or were colonized more recently tend to be in better condition (e.g. Galápagos Islands, Raine Island, Heron Island). Likewise, subpopulations in areas where turtle consumption has not been an integral part of the culture have been impacted to a lesser extent than those located where turtles or eggs have been a traditional food source. Rookeries in Australia, for example, have benefited from the fact that sea turtle consumption has not been an integral part of the dominant culture. A very different scenario is present in Southeast Asian countries, the Indian Ocean, and eastern Pacific Ocean (México), where green turtle subpopulations have suffered tremendously through harvest of eggs and turtles.

In respect to current conditions affecting green turtles, conservation programs have had a positive impact on nesting population trends around the world. In some cases conservation practices have enabled nesting subpopulations to partially or fully rebound from prior exploitation-induced declines. As a consequence of the slow maturation of green turtles, it is apparent that on-the-ground conservation programs must be in place for extended durations to reverse declines. Once exploitation threats have been eradicated, the recovery time of a subpopulation will depend largely on the status of the immature cohorts: subpopulations with a healthy immature stock will typically exhibit signs of recovery at the nesting beach more quickly than subpopulations with depleted immature stocks (Mortimer 1991, Crouse 2000).

In summary, regional differences in subpopulation trends are evident among the 32 Index Sites examined in this assessment. These differences are due to both the varying duration of exploitation and the history and quality of conservation programs in each region. Although this IUCN Red List assessment focuses on global status, the presence of regional subpopulation trends suggests that it is appropriate to apply the IUCN Red List Criteria at regional levels (Gärdenfors et al. 2001).

The Shifting Baseline Syndrome

Although extrapolations as per the IUCN Guidelines (IUCN 2001b) have provided some understanding about the historic subpopulation sizes at the 32 Index nesting rookeries, assessments of how today's subpopulations compare to those from pre-exploitation years may be erroneous. In several cases, perceptions suffer from the 'shifting baseline syndrome' (Pauly 1995). This situation arises when the greatest rates of decline take place prior to the earliest period for which subpopulation abundance data are available. As a result, subpopulations may be falsely classified as *stable* or *increased* when they are in fact depleted relative to historic levels. For example, the numbers of nesting females at Tortuguero, the most important rookery in the Caribbean, have increased since the onset of census counts in the early 1970s. When considering the exorbitant rate of extraction documented in other areas of the Caribbean over the last 141 years (for review see Groombridge and Luxmoore 1989, Fleming 2001) it is reasonable, however, to suspect that the nesting subpopulation at Tortuguero and other extant Caribbean rookeries were markedly larger 3-generations ago. Similarly, Ingle and Smith (1949), Parsons (1962), and Witzell (1994a) describe a Florida green turtle fishery that extracted a substantial number of turtles from Florida waters. In 1970 for example, the legal Florida green turtle harvest peaked at 190,013 kg (Witzell 1994b). Although there has been a steady increase in nesting numbers in this region over the past 20 years, current nesting activity likely represents only a fraction of historical levels. The fact that a shifting baseline may be resulting in the false perception of stable and increasing trends is underscored by estimates from Jackson (1997) that suggest the total adult green turtle population for the entire pre-Columbian Caribbean population ranged from 33 to 660 million turtles. Similarly, based on the assumption that Caribbean green turtle populations are regulated by the availability of turtlegrass (*Thalassia testudinum*), Bjorndal et al. (2000) estimated that between 16 and 568 million green turtles were present in the Caribbean prior to organized fisheries. These are rather wide intervals but even if historic green turtle population sizes were closer to the lower end of these ranges, the estimates would still represent a substantially greater number of green turtles than are present today.

The shifting baseline syndrome is widespread and variable in context. In addition to altering perceptions about the current stability of subpopulations, the shifting baseline syndrome may lessen the perceived intensity of historic declines for localities at which subpopulations are already classified as depleted. In Michoacán, México for instance, the population size in the early 1970s was estimated to be 25,000 nesting females per season (Cliffon et al. 1982). However, this was likely an estimate for an already depleted population, as green turtle harvests in the eastern Pacific Ocean had been ongoing for at least 50 years by that point (Averett 1920, Craig 1926). As noted by Carr (1961), the abundant green turtle populations were subjected to heavy extraction throughout the eastern Pacific Ocean for many decades. Speaking of the harvest at a single village in Baja California, México, Caldwell (1963) wrote, "I saw over 500 landed in a 3-week summer period in 1962 at Los Angeles Bay alone, and a comparable number, considering fishing effort, per week in winter". Extraction was so heavy that, during their investigations of green turtles, Caldwell and Caldwell (1962) coined this species the 'black steer' of the Gulf of California. When considering that Bahía de los Angeles was only one of many villages in northwestern México that had extensive fishing operations (Márquez and Doi 1973, Olguin Mena 1990), it is reasonable to believe that the combined efforts of these fisheries contributed to a significant decline in nesting numbers well before Cliffon et al.'s (1982) estimate.

The Shifting Baseline Syndrome also creates false perceptions about the degree to which some subpopulations have recovered from historic, exploitation-induced declines. At the Seychelles Islands in the western Indian Ocean for example, the annual nesting population has shown recent increases but remains depleted relative to historic numbers. The mean annual number of nesting females at Aldabra Atoll, presently the most important nesting area in the Seychelles, has increased since the onset of protective measures in 1968 (1984 – 1988 mean = 941 – 1,730; Mortimer 1988). However, upon expanding the temporal range of the inquiry, it is apparent that the present nesting level remains at least 72 % below that from the turn of the century (6,000 – 8,000; Mortimer, 1985).

A similar scenario is present in Sabah, Malaysia, where at least 40 years of intense egg harvest during the middle of the 1900s resulted in a highly depleted nesting population (de Silva 1982, Limpus 1995). In the early 1970s all three Sabah Turtle Islands were acquired and protected by the Sabah State Government (de Silva 1982a, b). Even with near total protection, the nesting stock continued to decline for nearly 20 years until the 1987/88 season when the subpopulation showed the initial signs of rebound. By 2001 nesting numbers exceeded those of the 1960s and it was proclaimed that the subpopulation had recovered (E. Chan pers. comm.). However, in 1965 (the year that served as the baseline for the recovery comparisons) the green turtle subpopulation had already been subjected to intense egg harvest since at least 1933 (de Silva 1982). It is reasonable to believe, based on current knowledge of the detrimental impacts of egg harvest, that the Sabah nesting subpopulation was considerably larger during the decades prior to 1965. Moreover, if the baseline were shifted farther back it would perhaps become evident that today's Sabah nesting subpopulation has yet to reach pre-exploitation levels.

Current and Future Threats

Despite improving conservation efforts throughout the world, green turtle subpopulations continue to be impacted by a variety of threats (Table 7). Four of the most common threats are intentional capture in foraging areas (16/32 Sites), incidental capture in marine fisheries (15/32 Sites), egg poaching (15/32 Sites), and disease (12/32 Sites). These impacts and others (summarized in Table 7) threaten the stability of many of the world's intact nesting subpopulations and also hinder recovery efforts for depleted subpopulations.

The adverse effects of egg harvest have been quantified and it is clear that long-term unregulated take is detrimental to sea turtle subpopulations (Chaloupka 2000). However, even if egg harvest is ceased, damage to the future stability of some stocks may have already been done. This is because the slow maturation rate of green turtles causes a delayed response to the negative effects of egg harvest. For example, when 100 % of the eggs are harvested from a nesting beach, reproduction is essentially halted: hatchling production ceases and no new juveniles recruit to the population. Without new recruits the population progressively loses important age classes that would normally replenish adults lost to natural or unnatural processes (Crouse 2000). After 30 years or more of egg harvest the true effects will be revealed when nesting numbers finally crash because all the juveniles have matured and can no longer offset the loss of adults (Mortimer 1995, Crouse 2000). The onset of such a crash may be exacerbated when nesting stocks experience a concurrent extraction of juveniles from in-water populations and/or unnatural rates of adult mortality. With regard to the sustainability of egg harvest, Chaloupka and Limpus (1998) warn that prolonged egg harvest, even for a period as short as 10 years, will have a major impact on stock viability. The probability of collapse is greatly enhanced when harvest is greater than 25% of the total egg production (Chaloupka 2000).

In addition to the collection of eggs from nesting beaches, there is a delayed response to the killing of nesting females. In areas where the killing of adults is commonplace, there may already be more than 30 years worth of juveniles in the marine environment, some of which mature and 'replace' the harvested adults each year. This replenishment prevents immediate declines in annual nesting numbers and creates a false perception of a population seemingly capable of sustaining adult harvests indefinitely (Crouse 2000). The killing of adults can also be considered a double negative in the sense that, not only are females being killed thus immediately reducing the population size, but with the loss of each female there is a corresponding decrease in egg production potential for the population.

Although adult mortality results in more quickly observable changes at the nesting beach, it is the mortality of immature turtles in marine habitats that may be of greater threat to the stability of green turtle subpopulations. Directed take of large juveniles is particularly worrisome as Crouse et al. (1987) and Frazer (in Ogren 1989) have identified this life-stage as the most valuable in terms of recovery and stabilization of sea turtle populations. This is due to the fact that not only have large juveniles already survived many mortality factors thus having a high reproductive value, but also there are typically more juveniles than adults in a population. Therefore, relatively small changes in the survival rate of this life-stage class impact a large segment of the population, magnifying the effect (Crouse 2000). As with the delayed feedback from egg harvest, green turtles' extended time to maturity delays the observable effects of juvenile harvests, and they may not manifest as a decline in nesting females for a number of decades. However, once there is a crash in the adult nesting population as a result of such impacts, the nesting population may be substantially more difficult to recover compared to a population with a thriving sub-adult population (Mortimer 1991). The fact that sea turtles exhibit fidelity to their natal beaches suggests these sites may not be replenished by the recruitment of turtles from other nesting rookeries in ecological time frames, a pattern consistent with the current distribution of nesting sites relative to human exploitation.

Green turtle populations in decline

Eighteen (56 %) of the 32 Index Sites examined in this assessment show declining trends in nesting activity. Many reductions are not only well in excess of 50 %, but also occurred in less than three generations. Thus, as demonstrated by the extrapolated declines (Table 5), the rate of these declines is much greater than 50 % over three generations. Based on published accounts, subpopulation declines of over 50 % have been identified in the eastern and western Atlantic Ocean; declines of greater than 80% have been shown in the eastern, southern, and western Pacific Ocean, Southeast Asia, Indian Ocean and Mediterranean Sea. Among all 19 declining subpopulations, the weighted mean rate of decline between documented *Past* and *Present* estimates (Table 4) is 62 % (SD = 24, range = 12 to 98 %) over a mean interval of 49 years (range = 7 y to 101 yr) or just over one generation, a rate much greater than 50% over three generations.

The most common intentional impact that has been documented among declining subpopulations is egg harvest: present at at least of eight of the 18 subpopulations in decline (Table 7). As previously discussed, this practice will cause further declines to these already depleted subpopulations. At the largest green turtle nesting beach along the Pacific Coast of México (Index Site #1), nearly all eggs were harvested for at least several decades prior to 1978, less than one generation ago (Cliffon et al. 1982). The problem persists today, albeit at substantially reduced levels (Alvarado et al. 2001). At the Berau Islands in Indonesia (Index Site

6), formerly one of the most important nesting areas in Indonesia, egg harvest continues at nearly 100 % (N. Pilcher, pers. comm.). As seen in other parts of Indonesia (Stringell et al. 2000), improvements in nesting activity through beach protection have been hindered by the closure of nesting beach conservation projects due to political turmoil (N. Pilcher pers. comm.). Throughout the Philippines, egg collection occurred at high levels until 1993 (Ramirez de Veyra 1994), and continues at > 50 % in some areas (Cruz 2002). At the Sarawak turtle islands of Malaysia (Index Site # 9), most of the eggs laid between 1927 and 1989 were harvested, certainly contributing to the 94% decline over three generations reported here; the harvest rate approached 100 % from 1963 to 1975 (Mortimer 1990). At Terengganu, Malaysia (Index Site # 10) nearly 97 % of all eggs were harvested between at least 1960 and 1988 (Hendrickson and Alfred 1961, Mortimer 1991). Egg collection continues at 58 % in this region (Chan pers. comm.). Although not quantified, egg harvest also continues at Bioko Island (Index Site # 25) in the western Atlantic Ocean (Fretey 2001).

In addition to egg harvesting, the take of nesting females continues to impact some subpopulations. Among the 18 Index subpopulations that are presently depleted, harvests of nesting females have likely contributed to declines at at least four Index Sites. Adult female harvest persists at various intensities: low level harvest of nesting adults has also been reported for subpopulations along the Pacific Coast of México (Alvarado et al. 2001) and the Seychelles Islands (Mortimer et al. 1996), while nearly 50 % of all nesting females are poached each year at Bioko Is. and mainland Equatorial Guinea (Tomas et al. 1999) and Sharma (PRD Yemen, N. Pilcher pers. comm.).

In regard to the capture of juveniles and adults in marine habitats, current practices are preventing the recovery of several depleted subpopulations. In Michoacán, México, despite more than 20 years of nesting beach protection (Alvarado and Delgado 2001) and a country-wide ban on sea turtle harvest (Anonymous 1990), recovery efforts have been hampered by the 5,000 to 10,000 turtles killed each year at neritic foraging habitats near Baja California, México (Seminoff 2000, Nichols 2001, Gardner and Nichols 2001). In the Indian Ocean, rookeries at Europa and Tromelin Islands that were stable until recently have begun to show signs of decline despite total protection at the nesting beaches (Table 4). This drop in annual nesting is due, at least in part, to green turtle fisheries in the eastern Indian Ocean, particularly in Madagascar: first reported in the 1990s (Rakotonirina and Cooke 1994, Mbindo 1996), this fishery currently lands thousands of green turtles each year (Andrew Cooke pers. comm. to J. Mortimer). When combined with incidental captures, green turtle mortality in the Madagascar region is believed to be at least 10,000 individuals each year (J. Mortimer pers. comm.). A similar situation has been described in the northern Indian Ocean in Oman, where, in 1990 for example, a combined 4,280 green turtles were taken through direct harvest (Salm 1991) and incidental capture (Hare 1991).

Stable and increasing green turtle subpopulations

Among the 32 Index Sites examined in this report, subpopulations are stable at three localities; Galápagos Islands (Ecuador), Karan Island (Saudi Arabia), and Ras al Hadd (Oman). Annual nesting populations have increased from the earliest documented abundance at 11 localities; Ascension Island, Bijagos Islands (Guinea-Bissau), Comoros Islands, Florida (United States), Hawaii (United States), Heron Island (Australia), Raine Island (Australia), Sabah Turtle Islands (Malaysia), Suriname, Tortuguero (Costa Rica), and Yucatan (México).

Although a substantial number of sites show stable and increasing trends, three points are worth noting that provide context for these non-declining patterns. First, it is likely that several

Sites are perceived as increasing due to the aforementioned Shifting Baseline Syndrome. If earlier data were available for these localities, it stands likely that the wider temporal interval would depict depletions relative to historic population sizes (see shifting baseline explanation above; e.g., Sabah). Second, several rookeries (e.g., Galápagos Is., Florida, Hawaii, Heron Island, Saudi Arabia,) represent small segments of the global meta-population. Although they may be showing encouraging trends, their small size relative to that of most declining subpopulations minimizes their effect when integrated into the global population trend. Third, a number of sites, including several of the largest rookeries, face increased threats of mortality in distant foraging habitats. Such impacts in the marine environment are likely to result in declining subpopulation trends at many of these localities in the future.

In Southeast Asia current evidence indicates that tens of thousands, perhaps more than 100,000 juvenile and adult green turtles are harvested annually (Pilcher 1999, Limpus et al. 2002). Many of these originate from Raine Island in the northern Great Barrier Reef, site of one of the world's largest congregation of nesting green turtles. The fact that a substantial number of juveniles are being killed suggests that the full impact from mortality in foraging areas has yet to be fully expressed in nesting abundance. Although turtles still nest in large numbers at this rookery, there has been a progressive decrease in the mean nesting size of females (Limpus et al. 2002). Limpus et al. (2002) suggest this is a response to a reduction in the proportion of older turtles to the population (probably due to mortality in Southeast Asia), and interpret this as an early warning signal that the northern GBR green turtle nesting population is in the early stages of decline. Moreover, in recent years there has been an upward trend in the mean remigration interval of turtles nesting at Raine Island (Limpus et al. 2002). Given that the remigration interval of females returning for only their second season is longer than that for turtles that have nested during multiple prior seasons (i.e. older turtles), the observed increase in mean remigration further supports the notion that fewer large turtles are present in the population (Limpus et al. 2002). It must be noted, however, that these trends could also be interpreted potentially good signs indicating a preponderance of new recruits to the population (N. Mrosovsky pers. comm., R. Márquez pers. comm.).

The green turtle subpopulation that nests in Sabah, Malaysia similarly faces impacts in distant foraging areas (Limpus 1995). Although this nesting subpopulation is currently in a recovering phase (E. Chan pers. comm.), sustained impacts in the marine environment jeopardize its long-term stability. Moreover, this subpopulation is very sensitive to large-scale environmental perturbations that may compound the adverse impacts in foraging areas (Chaloupka 2001). Imperfect nesting beach conservation strategies may also weaken stability of the Sabah rookery: artificially high incubation temperatures at the Sabah hatcheries are resulting in nearly 100 % female sex among hatchling stocks (Tiwoi and Cabanban 2000). This trend that may skew wild sex ratios and adversely affect future reproduction.

In the Caribbean, the aforementioned nesting colony at Tortuguero has exhibited encouraging trends over the last two decades; however, poaching of eggs and adults at the nesting beach and intense harvest of juveniles and adults at foraging habitats threatens the stability of this nesting subpopulation. From 1997 to 1999, despite beach protection efforts, a mean of 9.8 % of nests near the township of Tortuguero were poached (Troëng 2000) and, over the entire nesting beach, a mean of 600 adults were killed annually with a peak of 1,720 nesting adults poached in 1997 (Troëng 1998, Troëng and Rankin González 2000). Although poaching of nesting adults has been slowed in recent years, there are persistent impacts at foraging areas for this nesting subpopulation. For example, at the Miskito Cays along the Caribbean coast of Nicaragua, an

area considered to be the primary foraging habitat for turtles originating from Tortuguero, a mean of 9,357 turtles were killed per annum between 1994 and 1996 (Lagueux 1998). Legal harvest of green turtles continues in this area today (Fleming 2001, C. Lagueux pers. comm.). If this mortality is not ceased, or at least drastically reduced, the Tortuguero nesting subpopulation, which has long been considered the stronghold for green turtles in the Caribbean region, could experience substantial declines.

There have been steady improvements in the conservation of sea turtles throughout the world and these have likely contributed to the stable nature of the aforementioned rookeries. However, a stable nesting density is not necessarily indicative of a healthy subpopulation. While green turtles in Hawaii and Florida have demonstrated some encouraging signs of recovery after 20 years of protection efforts, the relatively recent increase in the incidence of Fibropapilloma disease threatens to eliminate improvements in the status of these stocks (Ehrhart 1991, Balazs and Pooley 1991, Balazs et al. 1992). The presence of this often-fatal affliction has increased significantly among stranded turtles over the past 20 years; increases in incidence during the past decade range from 47 % to 69 % in Hawaii (Murakawa et al. 2000). Green turtles studied near Molokai from 1982-1996 showed a massive increase in the disease over this period, peaking at 61% prevalence in 1995 (Murakawa et al. 2000). In Florida, the affliction rate reaches 62% in some areas (Schroeder et al. 1998). These sites have received considerable attention due to the incredibly high incidence of Fibropapillomatosis, but they are not the only subpopulations that are afflicted with this pathogen. Among the remaining 32 Index areas, Fibropapillomatosis has also been documented in Australia (Raine Island), Costa Rica (Tortuguero), Indonesia (Berau Islands), Malaysia (Sabah Turtle Islands), México (Yucatan Peninsula), Philippine Turtle Islands, Seychelles, and Venezuela (Aves Is.) [see Table 7 for citations].

The Remainder Category

In addition to the 32 Index Sites included in this report, there are many areas that host green turtle nesting for which there are no long term quantitative data. Green turtle nesting congregations of particular interest include, but are not limited to, those at the Aru Islands (Shultz 1984), western coast of Australia (R. Prince pers. comm.), Gulf of Carpentaria coast of Australia (C. Limpus pers. comm.), Pacific Coast of Costa Rica (Cornelius 1982), Natuna Islands (Schulz 1984), New Caledonia (C. Limpus pers. comm.), Papua New Guinea (Philip 2002), Scilly Atoll (Lebeau 1985) and additional islands of the South Pacific (C. Limpus pers. comm.). Despite the lack of quantitative data from these areas, the 'Remainder' category appears to be in overall decline. This conclusion is based on the overwhelming number of qualitative reports that describe declining green turtle subpopulations at non-Index areas (e.g., Groombridge and Luxmoore 1989, Salm and Humphrey 1996, Fretey 2001, Fleming 2001, C. Limpus pers. comm.). Although past versus present comparisons are not commonly possible, these reports suggest that green turtle declines have been extensive and widespread, occurring within one human generation. Declines are attributed to intentional harvest of eggs and adult females at nesting beaches, and juveniles and adults in marine habitats.

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information from within their respective regions (Table 9). Throughout the process the IUCN Marine Turtle Specialist Group Evaluators (Debby Crouse and F. Alberto Abreu-Grobois) provided invaluable assistance.

The information for this assessment has come primarily from published articles and inter-agency reports. In addition, a Green Turtle Status Questionnaire was drafted (in cooperation with F. Alberto Abreu-Grobois and Jeanne A. Mortimer) and distributed to individuals in over 40 countries. Questionnaires solicited information on subpopulation trends, past and present threats, and current conservation efforts. Information was augmented with interviews of several informants.

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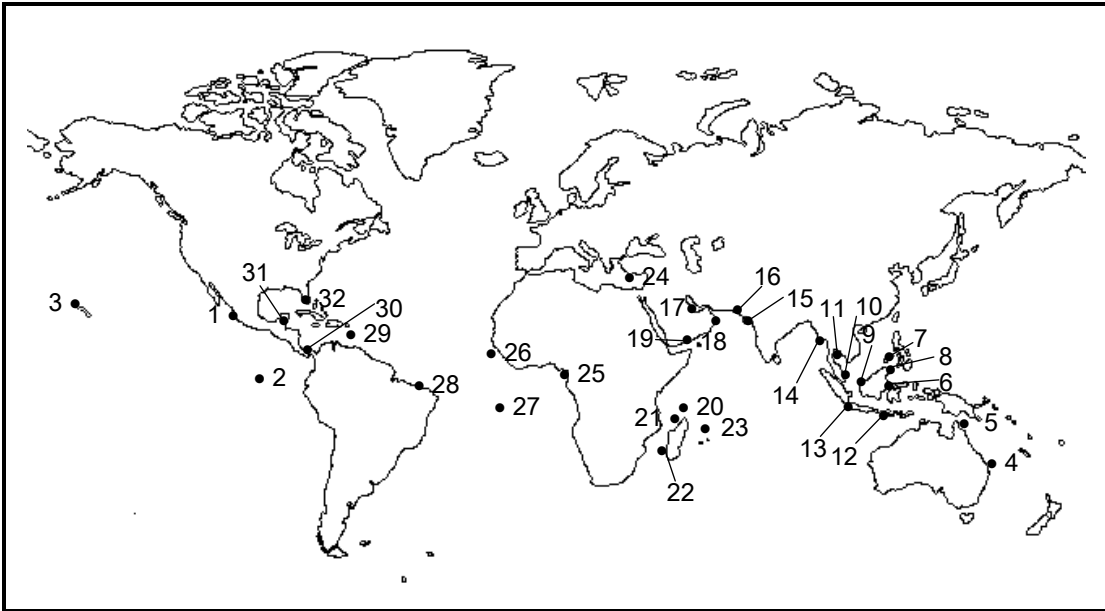


Figure 1. World map with the geographic locations of the 32 Index Sites used for the 2003 MTSG Green Turtle Assessment. See Table 1 for the rationale for inclusion of each site.

Table 1. Summary of 32 *Chelonia mydas* nesting rookeries used as Index Sites for the 2003 MTSO Global Green Turtle Status Assessment. See Figure 1 for map of all Sites.

Index Nesting Site		Justification
EASTERN PACIFIC OCEAN		
1.	México (Colola, Michoacán)	Historically the most important <i>C. mydas</i> nesting rookery in the eastern Pacific Ocean (Alvarado and Figueroa 1989).
2.	Ecuador (Galápagos Is.)	Currently the largest nesting congregation in eastern Pacific Ocean (Hurtado 1984, Hurtado 2001).
CENTRAL PACIFIC OCEAN		
3.	United States (Hawaii, French Frigate Shoals)	Hawaii has greatest nesting density of <i>C. mydas</i> in central Pacific; 90% of nesting in Hawaii is at French Frigate Shoals (Balazs 1980).
WESTERN PACIFIC OCEAN		
4.	Australia (southern Great Barrier Reef, Heron Is.)	Australia currently hosts some of the largest nesting congregations of green turtles in the world (Limpus et al. 2002); Heron Is. and Raine Is. represent the most important nesting areas in the sGBR and nGBR, respectively (Limpus et al. 2002).
5.	Australia (northern Great Barrier Reef, Raine Is.)	
SOUTHEAST ASIAN SEAS		
6.	Indonesia (Berau Islands)	Indonesia is among the most important nesting areas in the world (Groombridge and Luxmoore 1989); Berau Islands host some of the largest nesting colonies in Indonesia.
7.	Philippines (Turtle Islands)	Historically one of the largest and most important nesting colonies in Southeast Asia (Groombridge and Luxmoore 1989).
8.	Malaysia (Sabah Turtle Islands)	Historically important nesting colonies (de Silva 1982);
9.	Malaysia (Sarawak)	Sarawak and Sabah are two of the two most important insular regions in SEA; Terengganu has greatest nesting density in peninsular Malaysia (Mortimer 1991).
10.	Malaysia (Terengganu)	
11.	Thailand (Gulf of Thailand)	Increases area of coverage for SEA region.
EASTERN AND NORTHERN INDIAN OCEAN		
12.	Indonesia (Suka Made, Meru Betiri National Park)	Represents a nesting area in EIO that has been protected for several decades (Arrinal 1997)
13.	Indonesia (West Java, Pangumbahan)	Pangumbahan is most important nesting colony along the coast of Java (Groombridge and Luxmoore 1989).
14.	Myanmar (Thamihla Kyun)	Myanmar is a notable nesting area in northeast Indian Ocean region. Thamihla Kyun hosts largest nesting congregations in the area.
15.	India (Gujarat)	Provides added context for the Indian subcontinent.
16.	Pakistan (Hawkes Bay and Sandspit)	One of the largest nesting congregations along Indian subcontinent.
17.	Saudi Arabia (Karan Is.)	Largest nesting site in Arabian Gulf for which data are available.
18.	Oman (Ras al Hadd)	Historically one of the most important nesting areas in the northern Indian Ocean (Ross and Barwani 1982).
19.	Peoples Democratic Republic of Yemen (Sharma)	Described as "without any doubt one of the best nesting beaches remaining in the world" (Hirth and Carr 1970).

Table 1. - *Continued*

WESTERN INDIAN OCEAN		
20.	Seychelles Is. (Aldabra and Assumption)	Seychelles historically an important nesting area; Aldabra and Assumption represent two sites with largely different management histories.
21.	Comoros Islands	Currently one of the largest nesting rookeries in the western Indian Ocean.
22.	Isles Eparces (Europa Is.)	Europa Is. is a historically important nesting area in the western Indian Ocean and has total nesting beach protection.
23.	Isles Eparces (Tromelin Is.)	Tromelin Is. is one of the largest nesting congregations in the western Indian Ocean and has total nesting beach protection.
MEDITERRANEAN SEA		
24.	Turkey	Currently hosts the largest nesting congregation in the Mediterranean Sea (Kasperek et al. 2001).
EASTERN ATLANTIC OCEAN		
25.	Equatorial Guinea (Bioko Is.)	Important nesting area along the West African coast; Bioko Is. hosts almost all of nesting in this country (Groombridge and Luxmoore 1989).
26.	Guinea-Bissau (Bijagos Archipelago)	Guinea-Bissau currently hosts the largest nesting congregation along the West African coast (Fretey 2001).
CENTRAL ATLANTIC OCEAN		
27.	Ascension Is.	Represents the primary nesting rookery in the central Atlantic Ocean (Godley et al. 2001).
WESTERN ATLANTIC OCEAN		
28.	Suriname	Most important nesting area along northeastern South America.
29.	Venezuela (Aves Is.)	Presently the second largest rookery in the Wider Caribbean Region (Lagueux 2001).
30.	Costa Rica (Tortuguero)	Largest nesting rookery in the Caribbean Sea and intensively studied since 1956 (Carr et al. 1982, Bjorndal et al. 1999).
31.	México (Yucatan Peninsula)	Provides added context for the western Caribbean region. Includes the states of Campeche, Yucatán, and Quintana Roo.
32.	United States (Florida)	Provides added context for western Atlantic Ocean; only site included in southeastern United States.

Table 2. Estimated age-at-sexual-maturity^a for wild green turtles, *Chelonia mydas*. These published values are used in calculations of generation length for each Index subpopulation (see Table 3).

Study	Location	Age at	Reference
		maturity (years)	
A.	Hawaiian Archipelago	30	Zug et al. 2002
B.	Australia (nGBR)	30 ^b	Limpus and Walter 1980
C.	Australia (sGBR)	40	Limpus and Chaloupka 1997
D.	Florida	30	Mendonca 1981
E.	Florida	27	Frazer and Ehrhart 1985
F.	U.S. Virgin Islands	33	Frazer and Ladner 1986
G.	Ascension Island	35	Frazer and Ladner 1986
H.	Costa Rica	26	Frazer and Ladner 1986
I.	Surinam	36	Frazer and Ladner 1986

^a It has been suggested that a measure of mean nesting size will provide a closer estimate of the average size-at-maturity for green turtles than does minimum nesting size (e.g. Frazer and Ehrhart 1985, Limpus and Chaloupka 1997). Therefore, when possible, age-at-sexual-maturity is based on mean nesting size at each rookery.

^b Estimate based on minimum nesting size

NOTE: Additional growth data are available for subpopulations not listed in Table 2, however, these studies focused on head-started turtles (Ehrhart and Witham 1992, Burnett-Herkes et al. 1984), generated age-at-sexual-maturity estimates using un-reliable methods (e.g. Marquez and Doi 1973), or were based on non-applicable age classes (e.g. Zug and Glor 1998), thus reducing their utility for the present calculations.

Table 3. Summary of age-at-maturity, generation length, and calendar year of start date for Index subpopulations included in the 2003 MTSg green turtle assessment. See Table 2 for summary of the values used to determine age-at-maturity for each site.

#	Index Site	Age at Maturity (years)	Age at maturity calculation (From Table 2)	$\frac{1}{2}$ Reproductive Longevity (years)	Generation Length (GL; years)	3-generation duration ([= GL * 3]; years)	Calendar year 3 generations back (= 2001- 3GL)
1.	Eastern Pacific Ocean, México (Colola, Michoacán)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
2.	Eastern Pacific Ocean, Ecuador (Galápagos Is.)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
3.	Central Pacific Ocean, United States (Hawaii)	30	A	$\frac{1}{2}$ (19 yr) = 9.5	$30 + 9.5 = 39.5$	$39.5 * 3 = 118.5$	1883
4.	Western Pacific Ocean, Australia (sGBR, Heron Is.)	40	C	$\frac{1}{2}$ (19 yr) = 9.5	$40 + 9.5 = 49.5$	$49.5 * 3 = 148.5$	1853
5.	Western Pacific Ocean, Australia (nGBR, Raine Is.)	30	B	$\frac{1}{2}$ (19 yr) = 9.5	$30 + 9.5 = 39.5$	$39.5 * 3 = 118.5$	1883
6.	Southeast Asia, Indonesia (Berau Is.)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
7.	Southeast Asia, Turtle Islands, Philippines	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
8.	Southeast Asia, Turtle Islands, Malaysia (Sabah)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
9.	Southeast Asia, Malaysia (Sarawak)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
10.	Southeast Asia, Malaysia (Terengganu)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873

Table 3. – *Continued*

#	Index Site	Age at Maturity (years)	Age at maturity calculation (From Table 2)	$\frac{1}{2}$ Reproductive Longevity (years)	Generation Length (GL; years)	3-generation duration ([= GL * 3]; years)	Calendar year 3 generations back (= 2001- 3GL)
11.	Southeast Asia, Thailand (Gulf of Thailand)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
12.	Eastern Indian Ocean, Indonesia (E. Java, Suka Made)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
13.	Eastern Indian Ocean, Indonesia (W. Java; Pangumbahan)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
14.	Eastern Indian Ocean, Myanmar (Thamihla Kyun)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
15.	Northern Indian Ocean, India (Gujarat)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
16.	Northern Indian Ocean Pakistan (Hawkes Bay and Sandspit)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
17.	Northern Indian Ocean, Arabian Gulf Saudi Arabia (Karan Is.)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
18.	Northern Indian Ocean, Oman (Ras al Hadd)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
19.	Northern Indian Ocean, Peoples Democratic Republic of Yemen (Sharma)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873

Table 3. – *Continued*

#	Index Site	Age at Maturity (years)	Age at maturity calculation (From Table 2)	$\frac{1}{2}$ Reproductive Longevity (years)	Generation Length (GL; years)	3-generation duration ([= GL * 3]; years)	Calendar year 3 generations back (= 2001- 3GL)
20.	Western Indian Ocean, Seychelles (Assumption)	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
21.	Western Indian Ocean, Comoros Islands	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
22.	Western Indian Ocean, Isles Eparces, Europa	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
23.	Western Indian Ocean, Isles Eparces, Tromelin	33.3	Mean of A,B,C	$\frac{1}{2}$ (19 yr) = 9.5	$33.3 + 9.5 = 42.8$	$42.8 * 3 = 128.4$	1873
24.	Mediterranean Sea, Turkey	31.2	Mean of D,E,F,G,H,I	$\frac{1}{2}$ (19 yr) = 9.5	$31.2 + 9.5 = 40.7$	$40.7 * 3 = 122.1$	1879
25.	Eastern Atlantic Ocean, Equatorial Guinea (Bioko Is.)	31.2	Mean of D,E,F,G,H,I	$\frac{1}{2}$ (19 yr) = 9.5	$31.2 + 9.5 = 40.7$	$40.7 * 3 = 122.1$	1879
26.	Eastern Atlantic Ocean, Guinea-Bissau (Bijagos Archipelago)	31.2	Mean of D,E,F,G,H,I	$\frac{1}{2}$ (19 yr) = 9.5	$31.2 + 9.5 = 40.7$	$40.7 * 3 = 122.1$	1879
27.	Central Atlantic Ocean, Ascension Is.	35	H	$\frac{1}{2}$ (19 yr) = 9.5	$35 + 9.5 = 44.5$	$44.5 * 3 = 133.5$	1868
28.	Western Atlantic Ocean, Suriname (Galibi)	36	J	$\frac{1}{2}$ (19 yr) = 9.5	$36 + 9.5 = 45.5$	$45.5 * 3 = 136.5$	1865

Table 3. – *Continued*

#	Index Site	Age at Maturity (years)	Age at maturity calculation	$\frac{1}{2}$ Reproductive Longevity (years)	Generation Length (GL; years)	3-generation duration ([= GL * 3]; years)	Calendar year 3 generations back (= 2001- 3GL)
29.	Western Atlantic Ocean, Venezuela (Aves Is.)	31.2	Mean of D,E,F,G,H,I	$\frac{1}{2}$ (19 yr) = 9.5	$31.2 + 9.5 = 40.7$	$40.7 * 3 = 122.1$	1879
30.	Western Atlantic Ocean, Costa Rica (Tortuguero)	26	I	$\frac{1}{2}$ (19 yr) = 9.5	$26 + 9.5 = 35.5$	$35.5 * 3 = 106.5$	1895
31.	Western Atlantic Ocean, México (Yucatan Peninsula.)	31.2	Mean of D,E,F,G,H,I	$\frac{1}{2}$ (19 yr) = 9.5	$31.2 + 9.5 = 40.7$	$40.7 * 3 = 122.1$	1879
32.	Western Atlantic Ocean, United States (Florida)	29	Mean of D,E	$\frac{1}{2}$ (19 yr) = 9.5	$29 + 9.5 = 38.5$	$38.5 * 3 = 115.5$	1886

Table 4. Summary of published estimates of *Past* and *Present* nesting activity and subpopulation trends for *Chelonia mydas* at the 32 Index Sites. Data codes include: AN, nesting females; AC, number of nests; EP, egg production; EH, egg harvest; HP, hatchlings produced; and TC, tally count for high density nesting area. ALL VALUES ARE BASED ON ANNUAL MEANS UNLESS OTHERWISE STATED.

Index #	Subpopulation	Data type	Past Estimate 1		Past Estimate 2		Present Estimate		Citation (Past)	Citation (Present)
			Years	Mean	Years	Mean	Years	Mean		
1.	Eastern Pacific Ocean, México (Colola, Michoacán ^a)	AN	1970	15,000 females			1997-2001	851 females	Cliffon et al. 1982, R. Márquez, pers. comm.	Alvarado et al. 2001, R. Marquez, pers. comm.
2.	Eastern Pacific Ocean, Ecuador (Galápagos Is.)	AN	1976-1982	~1,400 females			1999-2001	~1,400 females	Hurtado 1984	Hurtado 2001, M. Hurtado, pers. comm.
3.	Central Pacific Ocean, United States (Hawaii)	AN	1974-1978	378 females			1991-2000	574 females	Balazs 1980, G.	Wetherall et al. 1998
4.	Western Pacific Ocean, Australia (Heron Is.)	AN	1964-1969	~400 females			1993-1999	562 females	Bustard 1974	Limpus et al. 2002
5.	Western Pacific Ocean, Australia (nGBR, Raine Is. ^b)	TC/ AN	1974-1979	2,361 females/night	1995-2000	3,680 fem/night	2001	~18,000 females/season	(1) Limpus et al. 2002; (2) Limpus et al. 2002	Dobbs 2002, K. Dobbs, pers. comm.
6.	Southeast Asia, Indonesia (Berau Islands, NE Kalimantan)	AN	1940s	~36,000 females; 200 fem/night, peak sea.			1984	~4000-5000 females; 25 fem/night, peak season	Schulz 1984	Schulz 1984
7.	Southeast Asia, Turtle Islands, Philippines	EH	1951	1,401,450 eggs			1981-1985	917,189 eggs	Domantay 1953, Groombridge and Luxmoore 1989	Reyes 1986 in Groombridge and Luxmoore 1989

Table 4. - *Continued*

	Subpopulation	Date type	Past Estimate 1		Past Estimate 2		Present Estimate		Citation (Past)	Citation (Present)
8.	Southeast Asia, Turtle Islands, Malaysia (Sabah)	EH/EP	1965-1968	556,278 eggs	1983-1986	255,877 eggs	1995-1999	975,480 eggs	(1) de Silva 1982; (2) de Silva in Groombridge and Luxmoore 1989; (3) Basintal and Lakim 1994, E. Chan, pers. comm.	E. Chan, pers. comm.
					Past Estimate 3					
					1989-1993	540,000 eggs				
9.	Southeast Asia, Malaysia (Sarawak)	EH	1927-1934	2,264,886 eggs	1981-1985	229,990 eggs;	1998-1999	228,618 eggs	(1) Banks 1937, Harrison 1962; (2) Leh 1986 in Groombridge and Luxmoore 1989	E. Chan, pers. comm.
10.	Southeast Asia, Malaysia (Terengganu)	EH, EP	1961	928,900 eggs	1993	317,105 eggs	1998-1999	218,354 eggs	(1) Hendrickson and Alfred 1961; (2) Ibrahim 1993	E. Chan, pers. comm.
11.	Southeast Asia, Thailand (Gulf of Thailand)	AC	1973-1983	405 nests			1992-2001	255 nests	Charuchinda and Monanunsap 1998	Charuchinda and Monanunsap 1998
12.	Eastern Indian Ocean, Indonesia (E. Java, Suka Made)	AC	1970-1974	1,555 nests			1991-1995	395 nests	Schulz 1987	Arrinal 1997, C. Limpus, pers. comm.
13.	Eastern Indian Ocean, Indonesia (W. Java; Pangumbahan)	EH	1950s	2,500,000 eggs			1980s	400,000 eggs	Schulz 1987	Schulz 1987
14.	Eastern Indian Ocean, Myanmar (Thamihla Kyun)	EH	1883-1898	1,744,164 eggs			1999	<250,000 eggs	Maxwell (1911) as cited in Groombridge and Luxmoore (1989)	Thorbjarnarson et al. 2000
15.	Northern Indian Ocean, India (Gujarat)	AC	1981	866 nests			2000	461 nests	Bhaskar 1984	W. Sunderraj, pers. comm.

Table 4. - *Continued*

#	Subpopulation	Data type	Past Estimate 1		Past Estimate 2		Present Estimate		Citation (Past)	Citation (Present)
16.	Northern Indian Ocean Pakistan (Hawkes Bay and Sandspit)	AC	1981-1985	1286 nests			1994-1997	~600 nests	Khan in Groombridge and Luxmoore 1989	Asrar 1999
17.	Northern Indian Ocean, Arabian Gulf Saudi Arabia (Karan Is.)	AN	1970s	500-1000 females			1990s	500-1000 females	Basson et al. 1977	Al-Merghani et al. 2000
18.	Northern Indian Ocean, Oman (Ras al Hadd)	AN	1977-1979	~6,000 females			1988	~6,000 females	Ross and Barwani 1982	Ross in Groombridge and Luxmoore 1989
19.	Northern Indian Ocean, Peoples Democratic Republic of Yemen (Sharma)	AN	1966, 1972	30-40 females/night, peak season			1999	750 females, 15 females/night, peak season	Hirth 1968, Hirth and Hollingworth 1973	Saad 1999
20.	Western Indian Ocean, Seychelles Islands ^c	AN	ca. 1900	11,000–13,000 females	1968	~1,700 females	1981-1984	3,535 – 4755 females	(1) Horneil 1927, Mortimer 1985; (2) Mortimer 1984, Mortimer 1988	J. Mortimer, pers. comm.
21.	Western Indian Ocean, Comoros Islands	AN	1972-1973	1,850 females			2000	5,000 females	Frazier et al. 1985	S. Ahamada, pers. comm.
22.	Western Indian Ocean, Isles Eparces, Europa ^d	AN	1970-1971	4-5,000; females	1978-1979	9-18,000 Females	1973-1985	2,000-11,000 females	(1) Hughes 1970; (2) Lebeau et al. 1983	Le Gall et al. 1986
22.	Western Indian Ocean, Isles Eparces, Europa ^d	HP	1983-1987	153,000 hatchlings			1990-1994	119,000 hatchlings	Rene and Roos 1996	Rene and Roos 1996
23.	Western Indian Ocean, Isles Eparces, Tromelin ^e	HP	1983-1987	427,600 hatchlings			1990-1994	377,000 hatchlings	Rene and Roos 1996	Rene and Roos 1996
24.	Mediterranean Sea, Turkey	AN	1978-1982	1,000 females			1998-2001	76-383 females	Geldiay 1987	Kasperek et al. 2001, Broderick et al. 2002

Table 4. - *Continued*

#	Subpopulation	Data type	Past Estimate 1		Past Estimate 2		Present Estimate		Citation (Past)	Citation (Present)
25.	Eastern Atlantic Ocean, Equatorial Guinea (Bioko Is.)	AH	1940s	200-300 females / night	1980s	50-100 females/night	1996/97-1997/98	1468 nests	(1) Eisentraut 1964; (2) J. Tomas, pers. comm.	Tomas et al. 1999
26.	Eastern Atlantic Ocean, Guinea-Bissau (Bijagos Archipelago)	AN	1990-1992	~2000 females			2000	~2465 females	Limoges and Robillard 1991, Paris and Agardy 1993 as cited in Fretey 2001	Catry et al. 2002
27.	Central Atlantic Ocean, Ascension Is.	AC	1977-1978	5257-10,764 nests			1998/99-2000/01	11,127 nests	Mortimer and Carr 1987	Godley et al. 2001, Broderick et al. 2001b
28.	Western Atlantic Ocean, Suriname (Galibi)	AN	1975-1979	1,657 females	1983-1987	1,740 females	1995	1,803 females	(1) Schulz 1982; (2) Mahadin in Ogren 1989	Weijerman et al. 1998
29.	Western Atlantic Ocean, Venezuela (Aves Is. ^f)	AN	1947	150-200 emergences /night, 1199 females	1984-1987	700-900 nests/season	1994	267 females	(1) Pinchon 1967 as cited in Pritchard and Trebbau 1984; (2) V. Vera pers. comm. to K. Eckert	V. Vera, pers. comm. to K. Eckert
30.	Western Atlantic Ocean, Costa Rica (Tortuguero)	AC	1971-1975	~41,250 nesting emergences			1992-1996	72,229 nesting emergences	Carr et al. 1982, modified from Bjorndal et al. 1999	modified from Bjorndal et al. 1999
31.	Western Atlantic Ocean, México (Yucatan Peninsula.)	AC	1983	~874 females			2000	~1547 females	Marquez 1984 a,b	R. Marquez, pers. comm.
32.	Western Atlantic Ocean, United States (Florida)	AN	1980	366 females			1995-2000	~759 females	Dodd 1982	Meylan et al. 1994, FMRI, INBDP (c/o B. Witherington)
	<i>Remainder^a</i>	AN				1860 – 2001		declining	Groombridge and Luxmoore 1989, Humphrey and Salm 1996, Fretey 2001, Fleming 2001	

^a Value for nesting females in Colola for 1970 is based on the estimate of 25,000 females for that year in all of Michoacán (Cliffon et al. 1982) multiplied by 60%, the relative amount of Michoacán nesting that is at Colola (R. Marquez, pers. comm.).

- ^b Dobbs (2002) estimates that the annual number of nesting females in nGBR is ~ 30,000. 60% of this is at Raine Is. (K. Dobbs pers. comm.)
- ^c Seychelles data are based on counts from Assumption Is. and Aldabra Is..
- ^d Two separate *Past – Present* data input lines are provided for Europa Island (Isles Eparces) to report (1) counts of nesting females and (2) hatchling production. Hatchling production data are based on the index site called Station Beach (M. Taquet pers. comm.) and represent only a subset of the entire production for Europa Is. Because these data more are based on hard counts rather than estimations presented in Ross (1982) we used them for the extrapolations in Table 3.
- ^e There are a variety of estimates available for Tromelin Island (see Hughes 1982), however the methods used to derive these estimates are unclear. Therefore, the present assessment is based on hatchling production data from the entire island (M. Taquet pers. comm.). Because these data are based on hard counts rather than unclearly derived estimations they were used for the extrapolations in Table 3.
- ^f At Aves Is., the Past estimate of nesting is based on estimate of 150-200 emergences per night during a one week period in 1947 (Pinchon 1967 as cited in Pritchard and Trebbau 1984). Taking this number and conservatively assuming that 1/3 of these were false crawls arrives at a nests/night estimate of 100-132. Using the midpoint of this estimate (116 nests/night) and, conservatively assuming that the season is only 1 month (31 d in July) long arrives at a value of 3,596 nests per season. At a rate of 3 nests per female, this equals 1199 females/season.
- ^g In addition to the 32 Index Sites included in this report, there are many areas that host green turtle nesting for which there are no long term quantitative data. Green turtle nesting congregations of particular interest include, but are not limited to, those at the Aru Islands (Shultz 1984), western coast of Australia (R. Prince pers. comm.), Gulf of Carpentaria coast of Australia (C. Limpus pers. comm.), Pacific Coast of Costa Rica (Cornelius 1982), Natuna Islands (Schulz 1984), New Caledonia (C. Limpus pers. comm.), Papua New Guinea (Philip 2002), Scilly Atoll (Lebeau 1985) and additional islands of the South Pacific (C. Limpus pers. comm.). Despite the lack of quantitative data from these areas, the 'Remainder' category appears to be in overall decline. This conclusion is based on the overwhelming number of qualitative reports that describe declining green turtle subpopulations at non-Index areas (e.g., Groombridge and Luxmoore 1989, Salm and Humphrey 1996, Fretey 2001, Fleming 2001, C. Limpus pers. comm.). Although past versus present comparisons are not commonly possible, these reports suggest that green turtle declines have been extensive and widespread, occurring within one human generation. Declines are attributed to intentional harvest of eggs and adult females at nesting beaches, and juveniles and adults in marine habitats.

Table 5. Summary of estimates of population change for the 32 green turtle Index Sites as determined with raw data, and Exponential and Linear extrapolation functions (IUCN 2001b). Past and Present published estimates are provided in Table 4. Subpopulation size units are mean annual number of nesting females. Unless otherwise stated, conversions from Table 2 data on number of eggs to number of nests and number of nests to number of females was determined using a mean value of 100 eggs/nest and 3 nests/female, respectively, for any given nesting season (Groombridge and Luxmoore 1989). Note: extrapolation functions are used only when there is a suspected change in the subpopulation size over a specific time interval outside of the period represented by data in Table 4. In such cases, unless otherwise noted, both linear (L) and exponential (E) functions are used due to a lack of information on the true rate of change over the time interval.

Index #	Subpopulation (Index Site)	Raw Data (from Table 4)		Notes on population trajectories	Past annual nesting female subpopulation size (3 generations back)	Present annual nesting female subpopulation size (2001)	% change
		Past	Present				
1.	Eastern Pacific Ocean, México (Colola, Michoacán)	15,000 (1970)	851 (1997-2001)	1873-1959: no change 1960-2001: declining (Craig 1926, Caldwell 1963, Seminoff et al. 2003).	37,851 (E)	851	- 98%
					19,564 (L)		- 96%
2.	Eastern Pacific Ocean, Ecuador (Galápagos Is.)	1,400 (1976-1982)	1,400 (1999-2001)	1873-2001: no change (Hurtado 1984, M. Hurtado, pers. comm.)	1,400	1,400	0%
3.	Central Pacific Ocean, United States (Hawaii)	378 (1974-1978)	574 (1991-2000)	1883-1977: no change 1978-2001: increasing (Balazs 1980, G. Balazs pers. comm.)	378	574 (E)	+ 52%
						583 (L)	+ 54%
4.	Western Pacific Ocean, Australia (Heron Is.)	400 (1964-1969)	562 (1993-1999)	1853-1968: no change (Parsons 1962) 1969-2001: increasing (C. Limpus pers. comm.)	400	575 (E)	+ 44%
						573 (L)	+ 43%
5.	Western Pacific Ocean, Australia (nGBR, Rainels ^a)	11,538 (1974-1979)	18,000 (1995-2001)	1883-1973: no change (MacGillivray 1910) 1974-2001: increasing (Limpus et al. 2002)	11,538	18,000	+ 56%

Table 5. - *Continued*

Index #	Subpopulation (Index Site)	Raw Data (from Table 4)		Notes on population trajectories	Past annual nesting female subpopulation size (3 generations back)	Present annual nesting female subpopulation size (2001)	% change
		Past	Present				
6.	Southeast Asia, Indonesia (Berau Islands)	36,000 (1940s)	4,500 (1984)	1873-1933: no change 1934-2001: declining (Schulz 1984, C. Limpus pers. comm.)	47,803 (E)	2,015 (E)	- 96%
				<i>Linear forward extrapolation would have resulted in a '0' estimate for present population size which is not possible. Therefore, only exponential forward extrapolation is used.</i>	40,295 (L)		- 95%
7.	Southeast Asia, Philippines ^b	4,886 (1951)	3,198 (1981-1985)	1873-1929: no change 1930-2001: declining (Domantay 1953, Reyes 1986).	6,348 (E)	2,620 (E)	- 59%
					5,929 (L)	2,404 (L)	- 59%
8.	Southeast Asia, Malaysia (Sabah ^c)	1,854 (1965-1968)	3,251 (1995-1999)	1873-1932: no change 1933-1986: declining (n ₁₉₈₆ =853) 1986-2001: increasing (de Silva 1969, 1982; E. Chan, pers. comm.).	8,389 (E)	3,994 (E)	- 52%
					3,800 (L)	3,620 (L)	- 05%
9.	Southeast Asia, Malaysia (Sarawak)	7,549 (1927-1934)	763 (1984-1988)	1873-1988: declining (Parsons 1962, Pelzer 1972, Mortimer 1990a); 1989-2001: no change (E. Chan pers. com.).	57,416 (E)	763	- 99%
					13,556 (L)		- 94%
10.	Southeast Asia, Peninsular Malaysia	3,096 (1961)	728 (1998-1999)	1873-1932: no change 1933-2001: declining (Hendrickson and Alfred 1961; C. Eng Heng, pers. comm.).	8,996 (E)	675 (E)	- 92%
					4,841 (L)	603 (L)	- 88%
11.	Southeast Asia, Thailand, Gulf of Thailand	135 (1973-1983)	85 (1992-2001)	1873-2001: declining (Parsons 1962, Charuchinda and Monanunsap 1998).	2,281 (E)	85	- 96%
					441 (L)		- 81%

Table 5. – *Continued*

Index #	Subpopulation (Index Site)	Raw Data (from Table 4)		Notes on population trajectories	Past annual nesting female subpopulation size (3 generations back)	Present annual nesting female subpopulation size (2001)	% change
		Past	Present				
12.	Eastern Indian Ocean, Indonesia (Suka Made, East Java)	518 (1970-1974)	132 (1991-1995)	1873-1949: no change 1950-1994: declining (Schulz 1984; C. Limpus, pers. comm.) 1995-2001: no change	2,471 (E)	132	- 95%
					959 (L)		- 86%
13.	Eastern Indian Ocean, Indonesia ^d (West Java)	8,333 (1950s)	1,333 (1980s)	1873-1949: no change 1950-2001: declining (Schulz 1984, Groombridge and Luxmoore 1989). <i>Linear forward extrapolation would have resulted in a '0' estimate for present population size which is not possible. Therefore, only exponential forward extrapolation is used.</i>	8,333	370 (E)	- 96%
14.	Eastern Indian Ocean, Myanmar	5,814 (1883-1898)	833 (1999)	1873-1882: no change 1883-2001: declining (Maxwell (1911) as cited in Groombridge and Luxmoore (1989), Thorbjarnarson et al. 2000).	7,759 (E)	802 (E)	- 90%
					6,554 (L)	734 (L)	- 89%
15.	Northern Indian Ocean, India (Gujarat)	289 (1981)	154 (2000)	1873-1966: no change 1967-2001: declining (Kar and Bhaskar 1982; W. Sunderraj pers. comm.)	460 (E)	149 (E)	- 68%
					388 (L)	147 (L)	- 38%
16.	Northern Indian Ocean, Pakistan (Hawkes Bay and Sandspit)	429 (1981-1985)	200 (1994-1997)	1873-1985: no change (Groombridge and Luxmoore 1989), 1986-2001: declining (Asrar 1999).	429 (E)	155 (E)	- 64%
					429 (L)	124 (L)	- 71%

Table 5. – *Continued*

Index #	Subpopulation (Index Site)	Raw Data (from Table 4)		Notes on population trajectories	Past annual nesting female subpopulation size (3 generations back)	Present annual nesting female subpopulation size (2001)	% change
		Past	Present				
17.	Northern Indian Ocean, Saudi Arabia (Karan Is.)	750 (1970s)	750 (1990s)	1873-2001: no change (N. Pilcher, pers. comm.).	750	750	0%
18.	Northern Indian Ocean, Oman (Ras al Hadd)	6,000 (1977-1979)	6000 (1988)	1873-2001: no change (Ross 1982).	6,000	6,000	0%
19.	Northern Indian Ocean, PRD Yemen (Sharma)	1,750 (1966-1972)	750 (1999)	1873-1949: no change 1950-2001: declining (Hirth 1968; Saad 1999).	3,490 (E)	704 (E)	- 80%
					2,565 (L)	676 (L)	- 74%
20.	Western Indian Ocean, Seychelles	12,000 (1900)	4145 (1981-1984)	1873-1899: no change 1900-1967: declining (Mortimer 1984) 1968-1884: increasing (N ₁₉₆₈ = 1,700) 1985-2001: no change (J. Mortimer pers. comm.).	12,000	4,145	- 65%
21.	Western Indian Ocean, Comoros Is.	1850 (1972-1973)	5000 (2000)	1873-1973: no change (Parsons 1962) 1974-2001: increasing (S. Ahamada, pers. comm.)	1,850	5,188 (E)	+ 180%
						5,117 (L)	+ 177%
22.	Western Indian Ocean, Isles Eparces (Europa Is. ^e)	463 (1983-1987)	360 (1990-1994)	1873-1987: no change 1988-2001: declining (Rene and Roos 1996, M. Taquet, pers. comm.).	463	280 (E)	- 40%
						257 (L)	- 44%

Table 5. – *Continued*

Index #	Subpopulation (Index Site)	Raw Data (from Table 4)		Notes on population trajectories	Past annual nesting female subpopulation size (3 generations back)	Present annual nesting female subpopulation size (2001)	% change
		Past	Present				
23.	Western Indian Ocean, Isles Eparces (Tromelin Is. ^f)	1,639 (1983-1987)	1,445 (1990-1994)	1873-1987: no change 1988-2001: declining (Rene and Roos 1996, M. Taquet, pers. comm.).	1,639	1,274 (E) ----- 1,251 (L)	- 22% ----- - 24%
24.	Mediterranean Sea, Turkey	1,000 (1978-1982)	230 (1998-2001)	1879-1919: no change 1920-2001: declining (Sella 1982, Kasperek et al. 2001) <i>Exponential backwards extrapolation would have resulted in an estimate for past population size which is not possible. Therefore, only linear backward extrapolation is used.</i>	3,513 (L)	230	- 93%
25.	Eastern Atlantic Ocean, Equatorial Guinea (Bioko Is. ^g)	2,075 (1940)	489 (1996-1998)	1879-1939: no change 1940-2001: declining (T. Butynski pers. comm. to K. Bjorndal as cited in Fretey 2001).	2,075	454 (E) ----- 407 (L)	- 78% ----- - 80%
26.	Eastern Atlantic Ocean, Guinea-Bissau (Bijagos Is.)	2,000 (1990-1992)	2,465 (2000)	1879-1989: no change (but see Agardy 1992) 1990-2001: increasing (Agardy 1992, Catry et al. 2002).	1,898 (E) ----- 1,884 (L)	2,530 (E) ----- 2,523 (L)	+ 33% ----- + 34%
27.	Central Atlantic Ocean, Ascension Island	2670 (1977-1978)	3,709 (1998-2001)	1868-1977: no change (but see Parsons 1962) 1978-2001: increasing (Godley et al. 2001).	2,670	3,709	+ 39%
28.	Western Atlantic Ocean, Suriname	1,657 (1975-1979)	1,771 (1983-1995)	1865-1978: no change 1979-2001: increasing (H. Reichart pers. comm.).	1,657	1,816 (E) ----- 1,814 (L)	+ 10% ----- + 10%

Table 5. – *Continued*

Index #	Subpopulation (Index Site)	Past	Present	Notes on population trajectories	Past annual nesting female subpopulation size (3 generations back)	Present annual nesting female subpopulation size (2001)	% change
29.	Western Atlantic Ocean, Venezuela (Aves Is.)	1,199 (1947)	267 (1979-1997)	1879-1899: no change 1900-1978: declining (Parsons 1962) 1979-2001: no change (V. Vera pers. comm. to K. Eckert).	10,887 (E)	267	- 98%
					2,568 (L)		- 90%
30.	Western Atlantic Ocean, Costa Rica (Tortuguero)	13,750 (1971-1975)	24,076 (1992-1996)	1895-1975: no change 1976-2001: increasing (Bjorndal et al. 1999, S. Troëng pers. comm.).	13,750	27,511 (E)	+ 100%
						26,535 (L)	+ 93%
31.	Western Atlantic Ocean, México (Yucatan Pen.)	874 (1983)	1,547 (2000)	1879-1982: no change (Parsons 1962) 1983-2001: increasing (R. Marquez, pers. comm.)	874	1,600 (E)	+ 83%
						1,587 (L)	+ 82%
32.	Western Atlantic Ocean, United States (Florida)	366 (1980)	759 (1995-2000)	1886-1979: no change (but see Parsons 1962, Witzell 1994a,b) 1980-2001: increasing (FMRI unpubl. data)	366	787 (E)	+ 115%
						779 (L)	+ 113%
TOTAL CHANGE USING RAW DATA + EXPONENTIAL FUNCTIONS					266,133	90,403	
TOTAL CHANGE USING RAW DATA + LINEAR FUNCTIONS					173,429	88,449	
<p><i>Therefore, the minimum change in global annual nesting female population size is from 173,429 to 90,403 females, or a 48% reduction and the maximum global annual nesting female population change is from 266,133 to 88,449 females, or a 67% reduction</i></p>							

^a For nGBR/Raine Island, past nesting numbers were determined by calculating the relative change in numbers of turtles observed during nightly surveys between Past and Present (+56%) and integrating this into current Raine Is. estimate of 18,000 annual nesting females (Dobbs 2002).

^b For Philippine Turtle Islands, conversion from egg data to # females was based on 95.6 eggs/nest (Trono 1991)

^c For Sabah, conversion from nest data to number of females was based on 5 nests/female/season (C. Limpus pers. comm.)

^d For Pangumbahan, Indonesia, conversion from egg data to # females was based on 107 eggs/nest (Suwelo and Kuntjoro 1969)

^e For Europa Is. conversions from hatchlings to number of females was determined using hatchling survivorship value of 77.6%; number of nests were determined using a value of 142 eggs/nest (Hughes 1974). Conversion to females from nests was based on a value of 3 nests per season per female (Bonnet et al. 1985).

^f For Tromelin Is. conversions from hatchlings to number of females was determined using hatchling survivorship value of 69.8%; number of nests

were determined using a value of 124.6 eggs/nest (Hughes 1974). Conversion to females from nests was based on a value of 3 nests per season per nesting female (Bonnet et al. 1985).

^g For Bioko Is. 1940 nesting subpopulation size (2,075) is based on a linear decline in turtles per night between 1940 (250 turtles) and 1980 (75 turtles) (=1.25 % / yr) extrapolated to 1998 (= 58.5 turtles/night). The actual turtles per season in 1998 (489) is then divided by this value to get a value for the number of turtles per season represented by each turtle counted in a night (= 8.3). This value is then multiplied by mean nightly count from 1940 (8.3 * 250) to get an estimate of the annual nesting N in 1940.

^h For Seychelles, the value for past nesters is based on the sum from Aldabra (5,000) + mid-point of range from Assumption (7,000). Value for present nesters is based on data provided in Mortimer (1984) and J. Mortimer pers. com

Table 6. Population changes for the 32 green turtle Index Sites grouped by region. Changes are determined with raw data and *Exponential (E)* and *Linear (L)* extrapolation functions as noted in Table 5. Past and Present published estimates and citations are provided in Table 4.

Region	Past annual nesting female subpopulation size (3 generations back)	Present annual nesting female subpopulation size (2001)	% change
EASTERN PACIFIC OCEAN (Colola, Mexico; Galapagos Is., Ecuador)	20,964 – 39,251	2251	-89% to -94%
CENTRAL PACIFIC OCEAN (Hawaii, USA)	378	574	+52%
WESTERN PACIFIC OCEAN (Heron Is., Australia; Raine Is. Australia)	11938	18573	+56%
SOUTHEAST ASIAN SEAS (Berau Islands, Indonesia; Turtle Islands, Philippines; Turtle Islands, Malaysia; Sarawak, Malaysia; Terengganu, Malaysia; Gulf of Thailand, Thailand)	68,862 – 131,232	9,490 – 10,151	-85% to -93%
EASTERN INDIAN OCEAN (Suka Made, Indonesia; Pangumbahan, Indonesia; Thamihla Kyun, Myanmar; Gujarat, India)	15,846 – 18,563	1,236 – 1,303	-92% to -93%
NORTHERN INDIAN OCEAN (Hawksbay and Sandspit, Pakistan; Karan Is., Saudi Arabia; Ras al Hadd, Oman; Sharma, Peoples Democratic Republic of Yemen)	10,132 – 11,129	7,696 – 7,758	-23% to -31%
WESTERN INDIAN OCEAN (Seychelles Is.; Comoros Is.; Europa and Tromelin, Eparces Is.)	15,952	10,770 – 10,886	-32%
MEDITERRANEAN SEA (Turkey)	3,513	230	-93%
EASTERN ATLANTIC OCEAN (Bioko Is., Equatorial Guinea; Bijagos Is., Guinea-Bissau)	3,959 – 3,973	2,930 – 2,984	-25% to -26%
CENTRAL ATLANTIC OCEAN (Ascension Is.)	2,670	3,709	+39%
WESTERN ATLANTIC OCEAN AND CARIBBEAN SEA (Galibi, Suriname; Aves Is., Venezuela; Tortuguero, Costa Rica; Yucatan Peninsula, Mexico; Florida, United States)	19,215 – 27,534	30,981 – 31,981	+13% to +66%

Table 7. Summary of recent and current threats documented for each of the 32 *Chelonia mydas* Index Sites included in the 2004 IUCN Red List Programme Species assessment. Presence of threats are indicated as Y = yes; N = no; ? = unknown.

Index #	Subpopulation	Existing Threats							Citations
		Intentional Take			Incidental Impacts				
		Egg Collection	Harvest of Nesting Females	Intentional Capture at Feeding Areas	Incidental Capture in Marine Fisheries	Habitat Loss at Nesting Beach	Contamination at the nesting beach	Disease	
1.	Eastern Pacific Ocean, México (Michoacán)	Y (1)	Y (1)	Y (2,3,4)	Y (3,4,5)	Y (1)	?	?	1. Alvarado-Díaz et al. 2001 2. Seminoff 2000 3. Nichols 2001 4. Gardner and Nichols 2001 5. NMFS 2001 6. Hurtado 2001 7. Balazs et al. 1998 8. Chaloupka and Limpus 2001 9. Limpus et al. 2001 10. Limpus et al. 2002 11. Limpus and Miller 1990 12. Dermawan 2002 13. C. Hitipeuw pers. comm. 14. Adnyana et al. 1997 15. Cruz 2002 16. Nalo-Ochona 2000
2.	Eastern Pacific Ocean, Ecuador (Galápagos Is.)	N (6)	N (6)	?	?	N (6)	N (6)	?	
3.	Central Pacific Ocean, United States (Hawaii)	N (5)	N (5)	?	Y (5)	?	?	Y (7)	
4.	Western Pacific Ocean, Australia (sGBR, Heron Is.)	N (8)	N (8)	N (8)	N (8)	?	?	?	
5.	Western Pacific Ocean, Australia (nGBR, Raine Is.)	?	?	Y (9)	Y (10)	?	?	Y (11)	
6.	Southeast Asia, Indonesia (Derawan (Berau Islands)	Y (12)	?	Y (13)	?	?	?	Y (14)	
7.	Southeast Asia, Philippines (Turtle Islands)	Y (15)	?	?	Y (15)	?	?	Y (16)	

Table 7. *Continued*

	Subpopulation	Egg Collect	Female Harvest	Intent. Capture	Incident. Capture	Habitat Loss	Cont.	Dis.	Citations
8.	Southeast Asia, Malaysia (Sabah Turtle Islands)	Y (17)	N (18)	?	Y (17)	?	?	Y (17)	17. E. Chan pers. comm. 18. Chan et al. 1997 19. Charuchinda et al. 2002 20. C. Limpus pers. comm. 21. Hutabarat pers. comm. 22. Thorbjarnarson et al. 2000 23. W. Sunderraj pers. comm. 24. W. Sunderraj pers. comm. 25. Asrar 1999 26. Pilcher 2000 27. Al-Merghani et al. 2000 28. N. Pilcher pers. comm. 29. R. Baldwin pers. comm. 30. B. Ferreira pers. comm. 31. Saad 1999
9.	Southeast Asia, Malaysia (Sarawak)	N (17)	N (17)	?	?	N (17)	?	N (17)	
10.	Southeast Asia, Malaysia (Peninsula)	Y (18)	?	?	?	Y (17,18)	?	N (17)	
11.	Southeast Asia, Thailand, Gulf of Thailand	?	?	?	?	Y (19)	?	?	
12.	Eastern Indian Ocean, Indonesia (Java; Suka Made, Meru Betiri NP)	?	?	Y (20)	?	?	?	?	
13.	Southeast Asia, Indonesia (Java; Pangumbahan)	Y (21)	?	Y (15)	?	?	?	Y (15)	
14.	Eastern Indian Ocean, Myanmar (Thamihla Kyun)	Y (22)	?	?	?	?	?	?	
15.	India (Gujarat)	Y (23,24)	N (23,24)	?	?	Y (24)	Y (25)	?	
16.	Pakistan (Hawkes Bay and Sandspit)	?	?	?	Y (25)	Y (25)	Y (25)	?	
17.	Northern Indian Ocean, Arabian Gulf (Saudi Arabia, Karan Is)	Y (26,27)	N (26)	?	Y (28)	N (28)	?	?	
18.	Oman (Ras al Hadd)	Y (29)	?	Y (29,30)	Y (29)	?	?	?	
19.	Peoples Democratic Republic of Yemen	?	Y (28,31)	?	?	?	?	?	

Table 7. *Continued*

	Subpopulation	Egg Collect	Female Harvest	Intent. Capture	Incident. Capture	Habitat Loss	Cont.	Dis.	Citations
20.	Western Indian Ocean, Seychelles	N (32)	Y (32)	Y (33,34)	Y (34)	Y (32)	?	Y (34)	32. Mortimer et al. 1996 33. A. Cooke pers. comm. to J. Mortimer 34. J. Mortimer pers. comm. 35. Mohadji et al. 1996 36. Rene and Roos 1996 37. Nada 2001 38. Kasperek et al. 2001 39. Godley et al. 1996 40. Godley et al. 1998 41. Godley et al. 1999 42. Fretey 2001 43. Formia 1999 44. Tomas et al. 1999 45. J. Tomas pers. comm. 46. Barbosa et al. 1998 47. Fortes et al. 1998 48. Broderick et al. 2001 49. van Tienen et al. 2000 50. H. Reichart pers. comm. 51. Bass et al. 1998 52. Guada and Solé 2000 53. Tambiah 1994 54. Fallabrino et al. 2000 55. Solé and Azara 1998
21.	Comoros Islands	?	Y (35)	Y (35)	?	?	?	?	
22.	Western Indian Ocean, Isles Eparces (Europa Is.)	N (36)	N (36)	Y (36)	Y (36)	?	?	?	
23.	Western Indian Ocean, Isles Eparces (Tromelin Is.)	N (36)	N (36)	Y (36)	Y (36)	?	?	?	
24.	Mediterranean Sea, Turkey	?	?	Y (37,38)	Y (38,39,40)	Y (38)	Y (41)	?	
25.	Eastern Atlantic Ocean, Equatorial Guinea (Bioko Is.)	Y (42)	Y (42,43,44)	Y (45)	?	?	?	Y (45)	
26.	Guinea-Bissau (Bijagos Archipelago)	Y (46)	Y (47)	Y (43,46)	Y (47)	?	?	?	
27.	Central Atlantic Ocean, Ascension Is.	N (48)	N (48)	?	?	N (48)	?	?	
28.	Western Atlantic Ocean, Suriname	Y (49,50)	N (49)	Y (50)	Y (50,51)	?	?	N (49)	
29.	Western Atlantic Ocean, Venezuela (Aves Is.)	N (52)	N (52)	Y (53,54)	?	?	?	Y (52,55)	

Table 7. *Continued*

	Subpopulation	Egg Collect	Female Harvest	Intent. Capture	Incident. Capture	Habitat Loss	Cont.	Dis.	Citations
30.	Western Atlantic Ocean, Costa Rica (Tortuguero)	Y (56)	Y (56)	Y (56,57)	?	N (56)	N (56)	Y (56)	56. Mangel et al. 2001 57. Lagueux 1998 58. K. Lopez pers. comm. 59. Schroeder and Mosier 2000 60. Ehrhart 1991 61. Balazs et al.1992
31.	México (Yucatan Peninsula)	Y (57)	Y (57)	?	Y (58)	Y (58)	?	Y (58)	
32.	United States (Florida)	?	?	?	?	Y (59)	?	Y (60,61)	
	Total number of Sites with threat present (Y)	15	8	16	15	9	3	12	
	Percent of Increased/Stable Sites with threat present	50% (7/14)	28% (4/14)	43% (6/14)	57% (8/14)	14% (2/14)	0% (0/14)	43% (6/14)	
	Percent of Declined Sites with threat present	44% (8/18)	22% (4/18)	55% (10/18)	39% (7/18)	39% (7/18)	17% (3/18)	33% (6/18)	

Table 8. List of Green Turtle Task Force Members for the 2001 IUCN Marine Turtle Specialist Group Assessment

Green Turtle Task Force Member	Affiliation / Institution
1. George H. Balazs	National Marine Fisheries Service, Honolulu, HI
2. Annette Broderick	Marine Turtle Research Group, University of Wales, Swansea
3. Karen Eckert	Wider Caribbean Sea Turtle Conservation Network
4. Angela Formia	Cardiff University, United Kingdom
5. Brendan Godley	Marine Turtle Research Group, University of Wales, Swansea
6. Mario Hurtado	Hurtado and Associates, Ecuador
7. Naoki Kamezaki	Sea Turtle Association of Japan, Osaka
8. Colin J. Limpus	Conservation Strategy Branch, Queensland Department of Environment & Heritage, Australia
9. Maria A. Marcovaldi	Fundacao Pro-TAMAR, Bahia, Brazil
10. Yoshimasa Matsuzawa	Sea Turtle Association of Japan, Osaka
11. Jeanne A. Mortimer	Ministry of Environment, Republic of Seychelles
12. Wallace J. Nichols	Wildcoast Conservation Team; California Academy of Sciences, San Francisco, United States
13. Nicolas J. Pilcher	University of Malaysia, Sarawak; Helen Reef project, Palau'
14. Kartik Shanker	Madras Consultancy Group, India

Table 9. List of Participants for the 2001 IUCN Green Turtle Assessment. Codes for type of information and assistance provided include: Q, Questionnaire submission; PC, personal communication/interview; R, provided published or unpublished reports; SC, provided helpful suggestions and comments, and; L, acted as liaison with additional contact persons.

	Name	Region	Type of Information
1.	Ridchard Adjei	Ghana	Q
2.	Said Ahamada	Comoros Islands	R
3.	Javier Alvarado	México	PC
4.	Windya Andana	Indonesia	Q
5.	Harry Andrews	India	Q
6.	Vincent Attard	Malta	Q
7.	Robert M. Baldwin	Oman	Q, PC
8.	Karen Bjorndal	Caribbean; Costa Rica	SC, PC, R, L
9.	Essô Bowessidjaou	Togo	Q
10.	Paulo Catry	Guinea Bissau	Q
11.	Milani Chaloupka	Australia	SC
12.	Mickmin Charuchinda	Thailand	R
13.	I-Jiunn Cheng	Taiwan	PC,R
14.	Jean-François Dontaine	São Tomé e Príncipe	Q
15.	Eng Heng Chan	Malaysia	PC, R, L
16.	Hamid Chfiri	Morocco	R
17.	Andreas Demetropoulos	Global Overview	SC
18.	Than Ngoc Diep	Vietnam	Q
19.	Josea Dossou-Bodjerou	Benin	Q
20.	Jacques Fretey	Africa	PC, R
21.	Betaina Ferreira	Spain	PC
22.	Nancy FitzSimmons	Southeast Asia	PC, R
23.	Matthew Godfrey	Global overview	SC
24.	José Gomez	Ivory Coast	Q
25.	Stephen S. Greeves	Liberia	Q
26.	Michael Griffin	Namibia	Q
27.	Hedely Guada	Venezuela	PC
28.	Hammou El Habouz	Morocco	R
29.	Mark Hamann	Australia	R, L
30.	Tran Minh Hien	Vietnam	Q
31.	Creusa Hitipeuw	Indonesia	Q
32.	Sahir Hussein	Maldives	Q
33.	Herda Hutabarat	Indonesia	Q
34.	Angoni Hyacinthe	Cameroon	Q
35.	Justus Joshua	India	Q
36.	Vijay Kumar	India	Q
37.	Max Kasperek	Mediterranean	R
38.	Cynthia Lagueux	Caribbean	PC, R
39.	Bojan Lazar	Adriatic Sea	Q
40.	Hock-Chark Liew	Malaysia	PC
41.	Karina Lopez	México	PC, R

Table 9. *Continued*

42.	Luis Felipe Lopez Jurado	Cape Verde	Q
43.	Sudharshani Kapurusinghe	Sri Lanka	Q
44.	Dimitris Margaritoulis	Mediterranean	R, L
45.	Rene Márquez	Global overview	PC, R, SC
46.	Amina Moumni	Morocco	Q
47.	Nicholas Mrosovsky	Canada	SC
48.	Samuel Kofi Nyame	Ghana	Q
49.	Joey Palma	Philippines	Q
50.	Bob Prince	Australia	Q, PC
51.	Peter Pritchard	Suriname	PC
52.	Tahir Qureshi	Pakistan	Q
53.	Henk Reichart	Suriname	Q, L
54.	Doinsoude Segniagbeto	Togo	Q
55.	Alhaji Siaka	Sierra Leon	Q
56.	Guy-Philippe Sounguet	Gabon	Q
57.	Thomas Stringell	Western Pacific	PC
58.	Hiroyuki Suganuma	Japan	PC,R
59.	S. F. Wesley Sunderraj	India	Q
60.	Pham Thouc	Vietnam	Q
61.	Manjula Tiwari	Morocco	Q
62.	Jesús Tomás Aguirre	Equatorial Guinea	Q
63.	Joca Thomé	Brazil	Q, L
64.	Sebastian Troeng	Costa Rica	PC, R, L
65.	Christopher John L. Ty	Philippines	Q
66.	Alessandra Vanzella-Khoury	Caribbean	R, L
67.	Blair Witherington	United States (Florida)	PC, R
68.	Ben Wolf	Nigeria	Q
69.	M. Abou Zaid	Egypt	Q
70.	H. Zahir	Maldives	Q
71.	Richard Zanre	Kenya	PC
