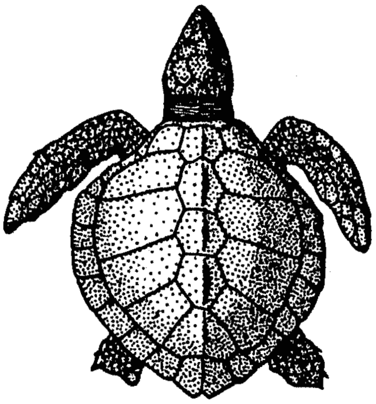
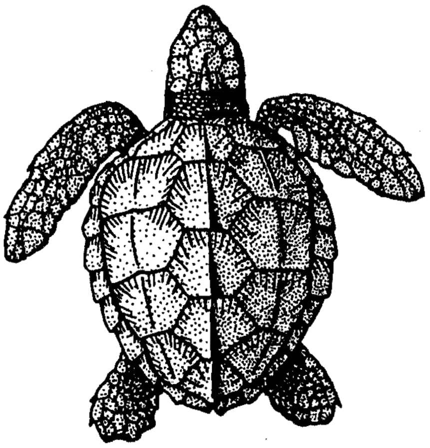
NOAA TECHNICAL MEMORANDUM
NMFS-SEFSC-409

AN ASSESSMENT OF THE KEMP'S RIDLEY (*Lepidochelys kempii*) AND LOGGERHEAD (*Caretta caretta*) SEA TURTLE POPULATIONS IN THE WESTERN NORTH ATLANTIC

A Report of the Turtle Expert Working Group





U.S. Department of Commerce
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Southeast Fisheries Science Center
75 Virginia Beach Drive
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March 1998

mortality rates of benthic immature and adult stages, and parameter estimates could be highly correlated. This correlation is as yet unevaluated.

Projection results are intended for descriptive rather than predictive purposes. The time horizon for these projections is long, given the short time scale for potential changes in anthropogenic mortality. Assumption of stable population dynamics processes over 20 years is unrealistic, especially when large changes in anthropogenically or environmentally induced mortality could occur rapidly and unpredictably. Moreover, the likelihood of obtaining trajectories similar to the ones presented here is liable to be small: a reasonable range of trajectory outcomes depends on uncertainty in starting stock size and parameter estimates.

Analysis of Trends - Strandings, Nests, and Hatchling Production

It may be difficult to distinguish trends in the number of stranded turtles from trends in population size (Crowder et al., 1995 (*SC strandings analysis*)). While the number of Kemp's ridley strandings increased in 1993, 1994 and 1995, the population may also have increased, given the large number of hatchlings released from Rancho Nuevo in preceding years and possible decreases in mortality due to TED implementation. We compared these trends by calculating the slopes of log-transformed strandings, hatchling production 2 years prior to each year's strandings (assumes a pelagic phase of 2 years), and Rancho Nuevo nest data (Table 1, Figure 1). We used the strandings data for all U.S. regions, 1986 - 1995, divided into the following categories: small benthic immatures (20 - 30 cm SCL), representing the youngest age class(es) present in benthic feeding habitats, benthic immatures (20 - 60 cm SCL), adults (> 60 cm SCL), and all benthic turtles (20+ cm SCL), which included all unmeasured animals. These four categories represented most of the turtles in the strandings record; pre-benthic (post-hatchling) sized turtles occasionally strand, but were excluded from this analysis. In general, strandings were far more variable from year to year than nests or hatchling production. The slopes, which provided the rate of increase in proportion per year for each data set, were highly dependent on which years were used. The TEWG has proposed that strandings and population trends be calculated over a 5-year interval for setting Interim Strandings Limits (ISL). The regression analysis indicates that from 1991 to 1995, benthic immature strandings and total benthic turtle strandings increased far more rapidly than population size, as indexed by nests or hatchling production. Adult strandings did not show a significant slope over this (or any other) time period; very few adult Kemp's ridleys strand in any one year, and this regression analysis was unable to detect a trend. The 1990 to 1995 analysis shows a much lower increase in benthic immature and total benthic turtle strandings, but the slopes were no longer significant at the $p < 0.05$ level. Over the entire data set (1986 - 1995), strandings have no significant slope. Analyzing the trend over this time period does not account for changes in mortality due to TED regulations; if TEDs are affecting mortality in the benthic immature and adult stages, we would anticipate a change in the strandings trend some time around 1990. From 1986 - 1990, none of the data measured showed a significant trend.

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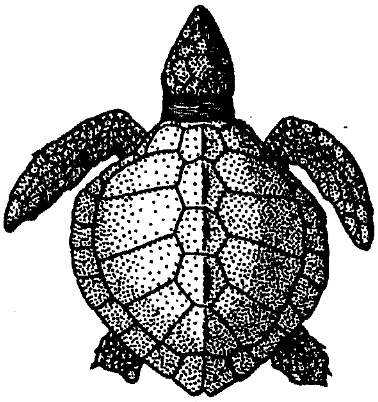

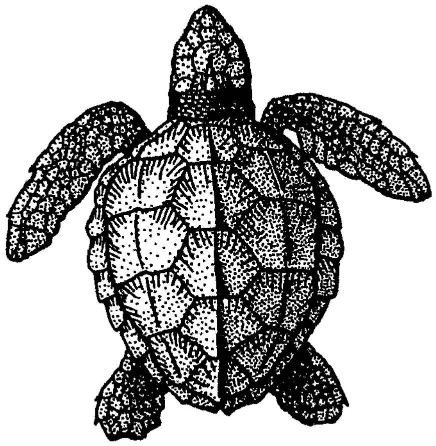

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A Report of the Turtle Expert Working Group

U.S. DEPARTMENT OF COMMERCE
William M. Daley, Secretary

NATIONAL OCEANIC AND ATMOSPHERIC ADMINISTRATION
D. James Baker, Administrator

NATIONAL MARINE FISHERIES SERVICE
Rolland A. Schmitten, Assistant Administrator for Fisheries

March 1998

NOTICE

This publication should be cited as follows:

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Preface

At the behest of NMFS, the SEFSC established the Turtle Expert Working Group (TEWG) in February 1995. The charge to the TEWG was first identified in the Endangered Species Act-Section 7 Consultation Biological Opinion for Shrimp Trawling in the southeastern United States under the Sea Turtle Conservation Regulations issued in November 1995. Specifically, NMFS was required to "select a team of population biologists, sea turtle scientists, and life history specialists to compile and examine formation on the status of sea turtle species. The team should attempt to identify a) the maximum number of individual sea turtles of each species that can be taken incidentally to commercial fishing activities without preventing the recovery of the species, b) the maximum number of individuals that can be taken incidentally to commercial fishing activities without jeopardizing the continued existence of any listed sea turtle species, and c) the number of stranded sea turtles occurring in statistical zone that indicate incidental takes are occurring at levels beyond those authorized." Further clarification was received in a letter of appointment from Mr. Rolland Schmitten, Assistant Administrator, NMFS/NOAA.

Since the first meeting in June, 1995, there have been 3 subsequent formal meetings which have culminated in the two stock level reports that constitute this publication. There is embodied in this publication a status report for the loggerhead turtle, *Caretta caretta*, within the southeast U.S. and the Kemp's ridley, *Lepidochelys kempii*, which is found in both Mexican and U.S. waters. This effort which represents the beginning of an evaluation of the status and condition of these stocks is considered preliminary and exploratory. We consider these efforts a work in progress. The TEWG is continuing in it's initial endeavors to develop methods to proceed with stock assessments for turtles and we are optimistic that this will be realized. It is the TEWG's desire that the results published in this report be interpreted with caution and within the bounds of the assumptions both with the data and with the methodologies applies. We wish to note that the exploratory modelling for the Kemp's ridley was completed only in the context of attempting to understand the life history of this species and to extract population level parameters which duplicate empirical data and numerical trends in abundance. This effort is considered a first step in exploring the use of stage or age specific models to assess the status of sea turtles and particularly progress towards recovery. Much hard work and deliberation has resulted in these reports to NOAA. I personally wish to acknowledge all the contributions of data and numerous late nights that were necessary to finalize this report. Considerable amounts of data were contributed that were not collected by NMFS and these cannot be used or duplicated without the written permission of the originator.

Nancy Thompson, Southeast Fisheries Science Center

Executive Summary

o The loggerhead stock within the southeast U.S. is subdivided into nesting assemblages based on genetic information. The extent that these represent separate stocks is not known. The South Florida Subpopulation which is currently the largest in terms of nesting females is stable and may be increasing; the status of the Florida Panhandle Subpopulation is unknown. Current trends for the Northern Subpopulation indicate this subpopulation has declined since the 1980's and that it is unlikely that the recovery goal will be achieved. The Yucatan Subpopulation may be decreasing but data are lacking for any time series for this assemblage.

o The Kemp's ridley population appears to be in the early stage of exponential expansion. While the number of females nesting annually is estimated to be orders of magnitude less than historical levels, the rate of increase in the annual number of nests has accelerated over the period 1986 to 1995. Under assumptions that estimated age to sexual maturity and age specific mortality rates are correct, preliminary analysis suggest that the intermediate recovery goal of 10,000 nesting females by the year 2020 is achievable.

o Empirical estimates of the number of turtles that can be removed from the population above natural mortality without preventing recovery cannot be made at this time with the data available. Stranding levels in the form of Interim Strandings Limits (ISLs) were developed for each species and are used as a proxy for mortality. Relationships between strandings levels, stock size, and mortality rates are unknown.

o ISL values were derived using a risk averse approach for both species based on 3 and 5 year averages of stranding totals for each species. For the loggerhead turtles 5 year running averages were estimated and one standard deviation was added to this average to provide the ISL. This approach recognizes the uncertainty of the status of this species and accounts for differences in the status between subpopulations. Both 3 and 5 year running averages were estimated for the Kemp's ridley plus one half the current level of increase observed in the annual number of nests. In this way, it is recognized that the rate of increase in strandings for this species should not be greater than the measured rate of increase of the population.

o Significant data gaps exist which limit the pursuit of complete age specific assessments for these species. Recommendations for research to obtain these data are included. Most notably for both species empirical estimates of age and survivorship are critical. Continued work to define subpopulations and rates of mixing on foraging areas of the loggerhead turtle need to be determined.

o Without a doubt current management which includes the use of TED's in the shrimp trawl fishery must be maintained for recovery of these stocks to be realized as defined by their Recovery Plans. Other sources of mortality must be empirically estimated and for fishery related mortality, the optimum method for estimation remains through the placement of observers on vessels.

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June 28, 1996
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STOCK DEFINITION AND GEOGRAPHIC RANGE

Kemp's ridleys (Lepidochelys kempii) occur in the Gulf of Mexico and northern half of the Atlantic Ocean (Pritchard 1989; Márquez 1994) and are assumed to constitute a single stock. The range of Kemp's ridleys includes the Gulf coasts of Mexico and the U.S., and the Atlantic coast of North America as far north as Nova Scotia and Newfoundland. They also have been reported from Bermuda, European Atlantic waters, the Mediterranean Sea, Madeira, the Azores and Nicaragua (Marquez 1994; USFWS and NMFS 1992).

Adult Kemp's ridleys are essentially restricted to the Gulf of Mexico (Chavez 1969; National Research Council 1990; USFWS and NMFS 1992). Adult-sized ridleys are occasionally found in Eastern U.S. waters but rarely nests outside the Gulf (i.e., in Florida, South Carolina and North Carolina, Meylan *et al* 1990; Palmatier 1993). Most Kemp's ridleys nest at the Rancho Nuevo nesting beach (Pritchard 1989). Pritchard and Marquez (1973) reported that adult females tagged at the nesting beach were captured by shrimp fishers in nearly equal proportions from the northern and southern Gulf. The distribution of adults appears to be mostly in the shallow, near shore zone (i.e., waters less than 50 m deep) as evidenced by satellite monitoring (Byles 1989; Renaud 1995).

Post-hatchling pelagic stages are assumed to become entrained in eddies within the Southwestern, Western and Eastern Gulf of Mexico (Collard 1990; Collard and Ogren 1990), where they are dispersed within the Gulf and Atlantic by oceanic surface currents until they reach about 20 cm in straight carapace length (SCL), at which size they enter coastal shallow water habitats (Ogren 1989). We define benthic immature Kemp's ridleys as those individuals of 20-60 cm straight carapace length (SCL) corresponding to the sizes generally found in the near shore habitat and not yet adults. Typically, they inhabit estuaries and adjacent near shore coastal waters (Ross *et al.* 1989; Rudloe *et al.* 1989; Epperly *et al.* 1992; Schmid and Ogren 1992; Epperly et al. 1995a, b). In the Gulf of Mexico, the habitat occupied by ridleys has been characterized by Hildebrand (1982) as white shrimp (*Penaeus setiferus*) and portunid crab habitats.

Some investigators believed Kemp's ridleys that exited the Gulf of Mexico were lost to the population, especially if they traveled to European Atlantic waters (Carr 1963, 1980; Hendrickson 1980; Brongersma 1982). However, others (Pritchard 1989; Collard and Ogren 1990; Schmid 1995) have concluded that the U.S. East coast is important developmental habitat for the benthic immature stages which return to the Gulf at some later period in their development. In 1994, a wild Kemp's ridley, tagged on the Southeast coast of Florida in 1989, nested at Rancho Nuevo (Schmid 1995). Prior to that time, the only indications of possible return to the Gulf of Mexico were seasonal north-south migrations along the Atlantic coast (Ogren 1989; Epperly et al. 1995a) and observations that the average size of Kemp's ridleys increases southward along the Eastern U.S. (Carr 1980; Henwood and Ogren 1987; see Epperly et al. 1995b for review).

STATUS

Kemp's ridley is the most seriously endangered of the sea turtles. Yet, its three to six million year existence (Bowen *et al.* 1991) shows it had been a very successful, well adapted species. Five decades ago, Kemp's ridley was very abundant in the Gulf of Mexico (USFWS and NMFS 1992). Its precipitous decline since 1947 (Figure 1) most likely was caused by human impacts at the primary nesting beach near Rancho Nuevo, Tamaulipas, Mexico and at sea[1] (Francis 1978; Hildebrand 1963; Chavez *et al.* 1968; National Research Council 1990; USFWS and NMFS 1992).

The Mexican government began protecting the Rancho Nuevo nesting beach from poachers in 1966, but the nesting population continued to decline (Chavez *et al.* 1968; Márquez *et al.* 1989; Márquez 1994). The U.S. Government listed Kemp's ridley as endangered in 1970 (USFWS and NMFS 1992), and the species received additional protection under the U.S. Endangered Species Act of 1973 (ESA), the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), and various laws, regulations, decrees and acts promulgated in Mexico (Márquez 1994). In 1978, U.S. agencies joined Mexico in protecting the nesting beach and in other efforts to prevent Kemp's ridley extinction (Woody 1986, 1989; Márquez 1994), but the number of nesters at Rancho Nuevo continued to decline by about 4% annually (Frazer 1986). In the mid-1980's, the number of Kemp's ridley nests laid at Rancho Nuevo began to increase in what now appears to be an exponential trend of 11% per year (Figure 1), but the species' population is still far below historical levels. The annual level of total human-caused mortality and serious injury (benthic immatures and adults) was estimated by the National Research Council (1990) to be 575 to 5,750 Kemp's ridleys.

The Kemp's ridley population appears to be in the early stage of exponential expansion. Due to the multi-year time lag between release of hatchlings and appearance of neophyte nesters of the same cohort (year-class), nesting beach management practices which enhance the annual production of hatchlings do not affect the annual number of nesting females or nests until years later. This time lag is determined by how long it takes the turtles to mature and nest for the first time, and it is not necessarily of the same duration for all surviving nesters of the same cohort. In contrast, management practices that reduce human-caused mortality and serious injury in turtles at sea can be expected to have both immediate and long-term effects which increase the annual number of nesters. Reductions in human-caused mortality and serious injury rates at sea (e.g., with Turtle Excluder Devices) can be expected to increase survival in all post-pelagic stages, assuming that natural mortality does not increase at the same rate as the decrease in human-caused mortality and serious injury.

Population Trends

The forty thousand nesting females estimated from a single mass nesting emergence (*arribada*) in 1947 reflected a much larger total number of nesting turtles in that year than exists today, perhaps by more than a 100 times (Carr 1963; Hildebrand 1963). For the two decades thereafter, a precipitous decline occurred in the Kemp's ridley population (National Research Council 1990; USFWS and NMFS 1992), during which production of hatchlings probably was low if not nil due to near total exploitation of eggs (Hildebrand 1963, 1982). Such exploitation certainly must have reduced, if not eliminated, recruitment of neophyte nesters to the beach at Rancho Nuevo for many years. The sharp decline in the Kemp's nesting population continued through the 1960s, as depicted by the decline in reported annual number of nests, and was followed by a decelerating decline to the minimum nests observed in 1985 (Figure 1). To a large extent, the annual number of nests reflects the number of nesting females, including new recruits (neophytes) to the nesting population. The decline in nests from 1966 through the mid 1980s can be explained by a decline in nesting females. There should have been a marked reduction in new recruits to the nesting population, based on the paucity of previous hatchling production. The decline in nesting females reflected both natural and anthropogenic mortalities, the latter of which predominately occurred at sea. The Kemp's Ridley Recovery Team (USFWS and NMFS 1992) attributed the decline to years of egg exploitation and the growth of the shrimp fishery post World War II. The brown shrimp (*Penaeus aztecus*) fishery in the Western Gulf was expanding during the period of decline in the Kemp's ridley population (Klima *et al.* 1982; Brunenmeister 1984; Rothschild and Brunenmeister 1984; Caillouet *et al.* 1991; Nance 1992).

The Kemp's ridley population decline might well have been irreversible had the Mexican government not begun to protect the nesting beach from poachers, beginning in 1966. When, in 1978, the U.S. Government began cooperating with Mexico in protecting the Rancho Nuevo beach, the annual number of hatchlings released more than doubled (Figure 2). The Magnuson Fisheries Conservation and Management Act of 1976 (MFCMA) extended U.S. fisheries jurisdiction to 200 NM offshore. Mexico extended its fisheries jurisdiction in 1978, thereby reducing shrimping effort in Mexican waters to the extent represented by exclusion of U.S. shrimpers.

Years 1987-1995 represent a period of accelerating increase in annual number of nests (Figure 1). We expect this upward trend in annual number of nests to continue. And it could accelerate, provided hatchling production and protection of sea turtles at sea are maintained.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Pursuant to Endangered Species Act Amendments of 1988, the National Research Council (1990) assessed the status of the Kemp's ridley sea turtle. Incidental capture in shrimp trawls was identified as the major cause of sea turtle mortality associated with human activities, killing more sea turtles than all other human activities combined. An estimated 500 to 5000 benthic immature and adult Kemp's ridley mortalities were attributed to shrimp trawling, prior to implementation of TED regulations. This compares to 75 to 750 estimated mortalities due to all other known human causes (NRC 1990).

Among the findings of the National Research Council (1990) were: (1) mortality of benthic immature and adult stages must be reduced to prevent extinction and effect recovery, under the condition that large numbers of hatchlings continue to be produced; (2) shrimp trawling kills more benthic immature and adult sea turtles than all other human causes combined; (3) in Texas and South Carolina, stranding increased with the onset of shrimping activities and decreased with the closure of Texas waters, indicating that 70-80% of the sea turtles stranding during the shrimping seasons were killed in shrimp trawls; (4) shrimping can be compatible with sea turtle conservation if adequate controls are placed on trawling activities, especially the mandatory use of TEDs at most times and places. Furthermore, the Kemp's Ridley Recovery Team (USFWS and NMFS 1992) concluded that incidental mortality caused primarily by commercial shrimping had forestalled recovery of the species by preventing adequate recruitment to the breeding population.

In addition to shrimp trawling, National Research Council (1990) identified many natural and anthropogenic causes of sea turtle mortality in waters and on the shores of the U.S. and Mexico including predation, parasitism, diseases, environmental changes, effects of beach manipulations on eggs and hatchlings, collisions with boats, entrapment in fishing nets and other gear, dredging, oil-rig removal, power plant entrainment, ingestion of plastics and toxic substances, and incidental capture in shrimp trawls. It is unlikely that any major non-shrimping cause of sea turtle mortality could have escaped detection during the many years of study of factors causing sea turtle injury and mortality at sea (National Research Council 1990, USFWS and NMFS 1992).

Federal regulations requiring TEDs in trawls used by offshore (seaward of the COLREGs demarcation line) shrimp trawlers longer than 25 ft were published in 1987 (Federal Register, Vol. 52, No. 124, p. 24247-24262, 28 June 1987). However, TED regulations were not implemented until May 1989, and TEDs were used only sporadically until May 1990 (McDonald 1990; Crouse *et al.* 1992; Henwood *et al.* 1992; Crouse 1993b; Weber *et al.* 1995). Year-round TED requirements became effective essentially in May 1991. After December 1992, year-around use of TEDs was required by most trawlers operating in Southeastern U.S. waters (57 FR 48861-48871, Sept. 8, 1992).

In the northwestern Gulf, Caillouet et al. (1996) found no change in the strength of the statistical association between nearshore shrimping intensity and sea turtle (all species) stranding rates in

TED years as compared to pre-TED years (Caillouet et al. 1991, 1996). Gallaway et al. (1995), following National Research Council (1990) methods, compared the reduction in strandings between the open and closed parts of the Texas shrimping season during pre-TED (1986-89) and post-TED years (1990-94). The magnitude in the reduction in strandings from the open to the closed period did not differ between post-TED and pre-TED years.

The Kemp's ridley population declined and remained very low during the early 1980s. Nesting stopped declining in the mid 1980s, before TEDs were required in U.S. shrimp trawls. The halt in the decline in the mid 1980s and the subsequent early stages of recovery of the Kemp's ridley population likely can be attributed to a number of factors, including protection of the nesting beach, associated enhancement of hatchling production, decrease in shrimp fishing effort in Mexican waters when the U.S. shrimping fleet was prohibited from shrimping in the late 1970s, and the overall decline in the Mexican shrimp fleet.

The shrimp fishery in the Gulf of Mexico

Threats to mating and nesting adult Kemp's ridleys from shrimping activity in Mexican waters have been decreasing. Shrimping effort off Rancho Nuevo was reportedly high until the U.S. and Mexican governments negotiated a bilateral agreement in 1976, phasing out U.S. shrimping in Mexican waters through 1979 (Iverson et al. 1993). Clearly, the reproductive component of the Kemp's ridley population was vulnerable to capture by this fishery. After 1979, U.S. vessels continued to shrimp off Mexico, although illegally and at reduced levels, through the mid 1980s, when the U.S. enforced the Lacey Act.

The reduction in effort in Mexican western Gulf waters caused by the departure of the U.S. fleet was furthered by the subsequent decline in the Mexican fleet. Currently, the entire Mexican fleet in the Gulf is estimated to number approximately 600 vessels, many of which are not presently fishing. Additionally, since 1978, waters out to 4 NM along approximately 14 km of the beach at Rancho Nuevo have been closed to fishing during the nesting season. However, this closure has not been strictly enforced, and until a few years ago, beach workers reported observations of ten to twenty trawlers operating from 2 to 5 mile off the beach at night. During 1995, Mexican waters of the Gulf closed to shrimping in concert with the Texas closure, from May 15 through July 15.

Significantly increased strandings, including a very high proportion of Kemp's ridleys, occurred throughout the southeast in 1994, but particularly in Texas and Louisiana. This was followed by elevated strandings again, but not as high, in 1995. Technical and intentional violations in gear installation, intense pulses of fishing at certain times and places, and use of TEDs with less effective at excluding turtles (NMFS, 1994) were all implicated as causes of the increased mortality. In addition, it is possible that an increase in the number of benthic immature Kemp's ridleys, due to the combined factors of increased hatchling production and TEDs in most US shrimp trawls, may be providing an increased pool of turtles.

Other sources of mortality include: Flounder fishing with otter trawls, off the coasts of North Carolina and Virginia, and gill netting for a number of species have also been implicated in high rates of sea turtle mortality on a more localized scale than shrimp trawling (National Research Council 1990). TEDs are currently required seasonally in all flounder trawls off North Carolina and Virginia and observers are documenting take rates for that fishery in the rest of its range.

POPULATION SIZE, VARIABILITY AND TRENDS

Kemp's ridley sea turtles provide a unique opportunity for population modeling because although they are endangered, we can monitor numbers of nests and hatchling output for nearly the whole population. Our approach, then, was to conduct statistical estimation procedures to estimate unknown parameters (e.g. stage-specific survival) for a preliminary population model. Other parameters were derived from empirical data and sensitivity analyses were conducted. Finally, population projections under various scenarios were conducted.

The primary indices of population status of Kemp's ridleys are currently limited to annual numbers of nests, observed nesting females, eggs and hatchlings produced at Rancho Nuevo (Márquez 1994), and size and imputed age composition and abundances of beach stranded turtles. The data include a 30-year time series of nests counted and hatchlings released (Table 1), and an empirical estimate of mortality rate for benthic immature turtles from a catch curve analysis of strandings (See 7 below).

The values we used as fixed parameters were: adult female remigration rate, the mean number of eggs per nest, the mean number of nests per female per year, female age at maturity, sex ratio in the population, and survivorship of benthic immature turtles. We considered published and unpublished information in developing the following values for the fixed parameters:

1. The **remigration rate of females** to the nesting beach was estimated from mark-recapture data (Marquez et al. 1982). We updated these data with current information provided by R. Marquez. Twenty percent of the turtles nest annually, 60% nest every 2 years, 15% nest every 3 years, and 5% nest every 4 years. This results in a mean remigration rate of 2.05 years, which we rounded to 2 years.

2. The mean **number of eggs/nest** incorporated into the calculations was 100.8 noted by the Kemp's Ridley Recovery Team (USFWS and NMFS 1992), which we truncated to 100 eggs/nest.

3. **Nests/female/season** were calculated by Rostal (1991) to be 3.075 from a physiological / ultrasonographic study. Pritchard (1990) estimated 2.31 nests/female/season and a recent pit-tag study by Marquez (unpublished data) suggests the mean is 1.8 nests/female/season. We used the mean of means from these studies (2.4 nests/female/season), which we rounded to 2.5 nests/female/season.

4. The **age at maturity** has been estimated by Marquez (1972) at 5-7 years. The Kemp's Ridley Recovery Team (USFWS and NMFS 1992) thought that estimate too low, but offered no better estimate. Zug et al. (1997) estimates Kemp's ridley maturity at 10 to 15 years based on analysis of cross-sections of humerus bones. He preferred the upper end of that estimate. From these data, we placed bounds of 7 to 15 years on age at maturity. Model runs were conducted with 8, 10 and 12 years input as the age to maturity parameter in order to test the model's sensitivity to the parameter.

5. A linear regression analysis was run to estimate **years in the pelagic stage** from hatchling production and number of stranded juveniles 1, 2, 3, and 4 years later in the stranding records. The analysis used stranding records for 20-30 cm, 30-40 cm, and 20-40 cm turtles. The resultant estimate of 2 years in the pelagic stage was used in the model.

6. **The sex ratio** in wild, benthic immature turtles, as determined by testosterone assay, has been found to approximate 1:1 at several locations throughout the U.S. (D. Owens, pers comm), but Stabeneau et al. (1996) found a sex ratio significantly skewed to females among stranded turtles. This female skew could be due to nesting females spending more time nearshore than males. We assumed a 1:1 sex ratio for this analysis.

7. **Instantaneous mortality rate, Z, for benthic immatures** was estimated using "catch curve" analysis of age composition derived by converting size distributions of post-pelagic stranded and measured Kemp's ridleys recorded in the Sea Turtle Stranding and Salvage Network (STSSN) database (1980-1995) to age distributions with a von Bertalanffy growth curve solved for age at size (see Sparre and Venema 1992). Curved carapace length (CCL) is the predominate measurement in the stranding database, so it was converted to straight carapace length (SCL), using a linear regression equation derived from stranded Kemp's ridleys for which both SCL and CCL were measured. When only SCL had been measured it was also included in the data analyzed. Size composition of post-pelagic stages fully recruited to the strandings was assumed to represent the size composition of the population of benthic immature and adult Kemp's ridleys. It was also assumed that growth of male and female Kemp's ridleys can be represented by the same growth curve.

Because of the scarcity of mark-recapture data for wild Kemp's ridleys, especially those in the benthic immature stage, mark-recapture data for head-started turtles (Caillouet *et al.* 1995*a*) were used to fit a von Bertalanffy growth curve in its ln-linear form, but SCL at infinite age was fixed at 75 cm instead of finding its value by iteration:

$$\ln[1 - (SCL_t/75)] = kt_0 - kt$$

None of the recaptured turtles had a SCL equal to or greater than 75 cm, so division into zero was not encountered in fitting the equation. The resulting von Bertalanffy equation was solved for t, as follows:

$$t = t_0 - (1/k)[\ln\{75 - SCL\}/75]$$

Since some stranded Kemp's ridleys had a SCL equal to or greater than 75 cm, whenever this condition was encountered, t was assigned a value of 15, because benthic immatures were assumed to fall well below 75 cm in SCL. Otherwise, t was estimated from SCL then grouped by class intervals into ages represented by integer values. For example, age 1 represented hatchlings and pelagic immatures less than 1 year old, age 2 represented pelagic immatures equal to or greater than 1 year old but less than 2 years old, age 3 represented benthic immatures equal to or greater than 2 years old but less than 3 years old, and so on. An age-frequency distribution was prepared for all year-classes. Year-classes were assigned by subtracting age from the calendar year. For example, for calendar year 1990, a turtle of age 1 year represented the 1989 year-class. The natural logarithm of the total number of turtles at each age 2-6 years was regressed on age to estimate Z for all year-class combined. This estimate of Z was 0.38 and was rounded to 0.4. It may have been biased by small sample sizes, pooling age distributions across year-classes, and the unknown relationship between strandings and total mortality.

8. We assumed **70% survival of eggs to hatchlings** based on recent survival rates in the corrals at Rancho Nuevo.

Parameter Estimation

In base model runs, the number of female hatchlings produced each year was entered as a starting cohort size, assuming a 50:50 sex ratio and no surviving hatchlings from other sources. Numbers of turtles in each age class in each year were projected from hatching production, assuming a two year pelagic phase with an associated mortality rate, and an eight-year benthic immature phase (in the case of age at sexual maturity of 10). Recruitment to the nesting population was partial in the first year of sexual maturity, determined by the remigration rate for the fully recruited population. Thus, at age of sexual maturity of 10, with a population remigration rate of 2 years, half of the 10 year-old turtles were added to the nesting population in the year they recruited. This assumption was required in order to scale the observed number of nests to the estimated nesting population, without knowing the percentage of new recruits in the population. The remaining adult population (e.g., age 11+ for age at sexual maturity of age 10) was modeled without additional age structure: number of nesting females at time t+1 was a function of number of adult females at time t and number of female recruits at time t, adult mortality rate, and remigration rate. The total number of predicted nests each year depended on constant assumed rates of nests produced per female, on the number of adults surviving from the preceding year, and the number of recruits to the nesting population.

Starting population sizes were derived from estimates of nests and hatchlings in 1966 (Table 1). The estimated number of nests in 1966 was converted to an estimate of nesting females in 1966 by applying rates of nests per female and remigration rate. That pool of nesting females was decremented by adult mortality rates between 1966 and the year the 1966 cohort recruited to the nesting population. This assumes that hatchling production before 1966, which would have supported recruitment to the nesting population in the intervening years, was effectively zero.

The model was updated on an annual basis, keyed to the reproductive rather than calendar year, i.e., estimates of stock size are as of the season of peak nesting rather than as of 1 January. Hatchling production was assumed to enter the population at that time as well.

Mortality rates of hatchling/pelagic turtles and adults, and a post-1989 decrease in benthic immature and adult mortality rates (see results) were estimated by comparing predicted nests to observed nests (1978-1995), using least-squares fit criteria (Excel 5.0 Solver^{TM} algorithm). We chose to model the 1978-1995 period because it included the most complete census of the nesting region.

Initial Model Runs

Initially, a single mortality rate per life history phase (hatchling, benthic immature and adult) was assumed to apply over the duration of the 1966-1995 time series. But inspection of residuals from a model estimating hatchling and adult mortality rates for the entire time series (combined with the fixed benthic immature mortality rate estimated empirically) revealed a strong trend in residuals from maximum negative values in 1989 to maximum positive values in 1995 (See Appendices). To address this problem, an additional term was fitted to the model as a single multiplier operating on both benthic immature and adult instantaneous total mortality rates during the 1990-1995 period. The period was chosen to reflect years in which TEDs were required. But it must be noted that the affect reflected in this term may arise from any combination of factors operating over the 1990-1995 period which in aggregate would decrease mortality rates, e.g., TEDs, favorable environmental factors, reduction in anthropogenic mortality from other sources, changes in fishery patterns driven by non-TED factors, etc.

Trends in residuals after inclusion of the post-1989 effect may indicate an increase between 1978-1984, with a possible decrease thereafter. This qualitative serial autocorrelation could be a direct result of the model formulation, in which number of nesters at time t+1 is a function of number of nesters at time t. It also could arise from the application of single stage-specific survival rates over the entire 1966-1989 time series, when more realistically, survival rates varied over time. Thus, residual patterns indicate a rough but not especially acceptable fit. Key diagnostic indicators of model performance such as matrices of correlations between parameter estimates were not considered for this analysis. Consequently, there is a significant possibility that too many parameters are being estimated given the model structure and the solution presented here may not be unique.

Mortality rates estimated from the model indicate a 44% annual proportionate survival rate of hatchlings during ages 0 and 1. This means that if 100,000 hatchlings were released, only 44,000 would reach age 1, and 19,400 (0.44 * 44,000) would reach age 2. In the benthic immature phase, proportionate survival rates of 66% were estimated from strandings data and input into the model. Of the 19,400 hatchlings reaching age 2, 12,800 would reach age 3 (0.66 * 19,400). With an age at sexual maturity of 10, turtles would remain in the benthic immature phase for 8 years (from ages 2 to 10). The 12,800 turtles at age 3 would decline to only 700 by age 10. Survival rate of sexually mature turtles was estimated at 84%. Thus, less than 600 turtles would remain by age 11, and less than 300 by age 15. With a 50:50 sex ratio, less than 350 females would survive from the initial 100,000 hatchings to reach sexual maturity, and fewer than 150 would be alive at age 15.

The survival rate for benthic immature and adult turtles was estimated to have increased substantially after 1989. The model was formulated so that lower initial survival rates would be increased faster than higher survival rates. The change in the rate was modelled as a multiplier of instantaneous mortality, (estimated as about a 33% reduction in instantaneous mortality). Instantaneous rates of mortality and post-1989 multipliers for those rates are reported in the Appendices. This meant that although mortality in the pelagic phase would continue during ages 0 and 1 at the same rate, survival rate of immature individuals was estimated to have increased to 75% and survival rate of adults was estimated to have increased to 89%. Thus, although 19,400 individuals would reach age 2, now 14,600 would reach age 3. Those 14,600 individuals would decline to 2,000 by age 10. With an adult survival rate of 89%, 1,800 turtles would remain by age 11, and 1,100 would remain by age 15. With a 50:50 sex ratio, less than 1,000 females would survive from the initial 100,000 hatchlings to reach sexual maturity, and perhaps 550 would be alive at age 15. Of those 550 to 1,000 females, half would nest each year.

It should be noted that although large numbers of turtles may occur at immature stages, (e.g., the 19,400 turtles at age 2 and the 14,600 turtles at age 3 in the example above) the number in a year class that survive to contribute to the nesting population in any year is small. Under recent improved survival conditions, 500 out of 100,000 hatchlings would contribute nests 10 years later, and only 225 would contribute 15 years later.

Based on model results, the abundance of benthic immature turtles (ages 2 - 9) has trended upward in three phases since 1966 (Appendix 1). Between 1966-1977, abundance initially increased fairly rapidly then leveled off, as hatchling production in 1966 began to contribute to the benthic immature population. Hatching production averaged about 23,000 per year between 1966-1977. A second period of increase and subsequent leveling off occurred between 1978-1989, as hatchling production trended upward over the 1976- 1979 period, averaging 49,000 per year between 1978-1987. The third period, from 1990 on, was one of steady increase. It began in 1990 as hatchling production increased from about 62,000 in 1988 to 84,000 in 1993, and as estimated survival rates of immature turtles increased as well. Abundance of benthic immature turtles in 1995 was estimated at 54,000 individuals, 55% of which were age 2 -3 and <2% of which were aged 9.

A qualitative relationship may emerge between estimated number of dead benthic immature animals and numbers of strandings from corresponding size categories (Appendix 1). Relatively uniform and widespread monitoring of strandings occurred after 1985, and indicates a variable pattern without trend between 1986-1993, but sharp increases in 1994-1995. Model estimates of numbers of dead benthic immature individuals were fairly stable between 1986 and 1991, and trended upward thereafter, as a function of increasing population size. Any statistical relationship would be driven by the co-occurrence of both maximum strandings levels and maximum estimates of deaths in 1994-1995. The increasing number of deaths estimated between 1991-1993 is not reflected in an increase in strandings during that period, however.

Abundance of adults declined from the population that produced about 6000 nests in 1966 to a population that produced 924 nests in 1978 and a low of 702 nests in 1985. The historical trajectory of adult abundance tracks trends in nest abundance, from an estimate of 9,600 in 1966 (from conversion of nests to population size, using ratios as described above) to 1050 in 1985 (Appendix 1). The adult population has increased to about 3,000 in 1995, consistent with 1,930 nests observed in 1995.

Based on model simulations, the percentage of neophyte nesters in the population increased steadily from just 13% to about 56% over the 1981-1989 period. The lower levels estimated for 1990 and 1993 (30%) are the result of low hatchling production in 1980 and 1983 projected to age at sexual maturity. Estimates for other recent years have been in excess of 40%. Direct observations of percentage of neophyte nesters from nesting beaches may be considered upper bounds (because some animals may not have been tagged during previous nesting seasons), and have ranged from 23- 28% in 1990-1991 to 38-41% in 1992-1994. There thus is fair correspondence between recent direct observations and model estimates.

Sensitivity Analyses: Number of nests in 1966

Because the estimated adult survival rate must be consistent with the decline in number of nesting females estimated from nests in 1966, and because number of nests in 1966 was an approximate estimate rather than a complete census, the effect of a +/- 25% difference in 1966 nest level was examined. The model was re- fitted using 1966 nest levels of 7500 and 4500 (Appendix 2, Appendix 3).

If the 1966 nest level were closer to 7500 nests (Appendix 2), estimated adult survival would be lower, to produce a steeper decline between 1966 and 1978. That steeper decline in the remnant nesting population after 1978 would be compensated for by a slightly increased contribution of neophytes in order to obtain nest production thereafter. The slight increase in contribution by neophytes would be driven by an increase in estimated survival rate of pelagic juveniles in earlier years of the recruiting cohorts' life. Lower increases in benthic immature and adult survival rates after 1989 would be required to match increased nest levels, because the standing stock of benthic immature individuals (and subsequent contributions to the nesting population) would be higher due to increased pelagic survival. The combination of higher pelagic juvenile survival rates with slightly lower survival rates of benthic immature and adult turtles would lead to a similar (albeit marginally smaller) recovery rate in future years. The residual sum of squares for this model was higher than the baseline run of 6000 nests (age at sexual maturity = 10).

Similarly, if the 1966 nest level were closer to 4500 nests (Appendix 3), adult survival would be higher and survival of pelagic juveniles would be correspondingly lower, with mechanisms exactly opposite from those described above. The combination of lower pelagic juvenile survival rates and higher benthic immature and adult survival rates would lead to a similar (albeit marginally larger) recovery rate in future years. The residual sum of squares for this model was lower than the baseline run of 6000 nests, and residual patterns showed less pronounced trends over time.

Sensitivity Analyses: Age at Sexual Maturity

Estimates of survival rates of benthic immature turtles from strandings data were independent of age at sexual maturity. The effect of increasing the age at sexual maturity (relative to the base level of 10) is to increase the number of years within the cohort's lifespan in which that rate is applicable. The aggregate mortality between pelagic juvenile and adult phase of a cohort is increased. The fitted results from this formulation indicate that the minimum residual sum of squares is obtained through an increase in pelagic juvenile survival, to compensate for aggregate decreased survival of benthic immature turtles over the longer immature period.

In a model formulation assuming age at sexual maturity of 12 years, estimated proportionate survival of pelagic immature turtles increased to 71% from 44% estimated for age 10. Estimated survival in the pelagic phase was higher than survival in the benthic immature phase, and on the same scale as survival in the adult phase (85%). The increased pelagic survival rate together with a greater number of contributing age classes led to greater abundance of benthic immature individuals. The increase in survival rate post-1989 was thus estimated to be lower. The percentage of neophytes in the nesting population was lower, as cohorts were decremented by additional years of mortality before recruiting to the nesting population. Recovery rates were similar to the base run for near future years, albeit marginally smaller. The residual sum of squares was almost three times that of the base level run (Appendix 4).

In a model formulation assuming age at sexual maturity of 14 years, estimated proportionate survival of pelagic immature turtles increased to the constraining upper bound of 100%, and unconstrained estimates of survival exceeded 100%.

If the estimate of survival of benthic immature turtles from strandings data is reasonable, then model results obtained by assuming age at sexual maturity of 12 years and greater appear unreasonable given the current model formulation. Other formulations, objective functions or estimation algorithms may give more acceptable results for older ages at maturity. Alternatively, if survival of benthic immature turtles were overestimated, meaningful results would be obtained from formulations based on older ages at sexual maturity.

Sensitivity Analysis - Benthic Immature Instantaneous Mortality

Because our estimate of the benthic immature instantaneous mortality rate may have been biased due to small sample sizes, inappropriate growth rate estimation procedures and the unknown relationship between strandings and total mortality, we examined the sensitivity of the 10 years to maturity model results to changes in benthic immature mortality. We ran the Solver™ algorithm with six different benthic immature mortality rates, using a range of benthic immature mortality rates that spanned our estimates from the strandings records. The program chose best fit parameters (adult instantaneous mortality, pelagic juvenile instantaneous mortality, and the post-1989 decrease in mortality) for each benthic immature mortality rate, with the reproductive parameters again held constant (Table 2). Adult survival remained constant at 0.84 for all sensitivity runs, primarily because we fixed the number of nests produced in 1966 at 6000. Pelagic juvenile mortality compensated for most of the change in benthic immature mortality; the negative correlation between these two mortality rates resulted in the same number of females recruiting to the nesting population for all models. Thus, the residual pattern remained nearly constant for each model, although there was a trend toward a better fit as the benthic immature mortality rate increased (Table 2). There was almost no change in time to reach 10,000 nesting females (Table 3). However, the estimated number of benthic immatures in the population was highly dependent on the allocation of mortality to the pelagic and benthic immature stages (Table 4). Higher annual survival rates for benthic immatures gave best fit models with lower survival for pelagic immatures, so many fewer survivors from a cohort actually recruited to the benthic immature stage, resulting in a smaller estimated population size. The opposite was true for lower benthic immature annual survival rates. Because we do not have empirical estimates for annual survival rates of any life stage, any of these best fit models could be closest to the actual mortality schedule of Kemp's ridleys. It is extremely tenuous to use this model to estimate the number of benthic immatures in the population.

Model Projections

Projected recovery trajectories were calculated from 1995 forward, based on 1995 standing stock size and mortality estimates obtained from the regression model assuming age at sexual maturity of 10. Future hatchling production also was modeled in density-independent and density-dependent forms. Limitations on human intervention and protection of Kemp's ridley nests were imposed on the model. Protection of the nests from predators at the Rancho Nuevo nesting beach is effort dependent, and will change with time. The biological desirability and logistics of protecting a large number of nests by transporting each nest to a corral (a fenced, natural sand hatchery on the beach) for incubation will likely limit the number of nests so protected. When some peak number of nests is corral-protected, the remaining nests will be protected in situ.

The INP and FWS have suggested that under current planning a maximum of 3,000 nests will probably be protected in corrals, with an additional 2,000 nests protected in situ by screening. Nests deposited beyond these 5,000 would be left completely natural although some method of predator control may become desirable. Hatch rates were assigned to the various protective regimes as follows:

# NESTS	CATEGORY	HATCH RATES (%)
0-3,000	Corrals	75
3,001-5,000	In situ: Screening	50
5,001-10,000	In situ: Low predation	30
>10,001	In situ: High predation	20

The density-independent form of the model was based on assumptions of 2.5 nests per nesting female per year, 100 eggs per nest and 75% survival of those eggs. The density-dependent form was based on similar assumptions of nests per female and eggs per nest, but allowed current levels of 75% egg survival to apply only to the first 3000 nests, the assumed overall level of protection obtainable through corral egg protection programs. The next 2000 nests would be protected with in situ techniques, with postulated egg survival rates of 50%. The next 5000 nests would be relatively unprotected, with egg survival dropping to a hypothetical 30%, under conditions of low predation mortality. At nest levels above 10,000, higher predation mortality would be obtained, with egg survival of a postulated 20% resulting. Trajectories are presented for the more realistic density-dependent case, although results in initial years of the projection are identical for the two mechanisms.

Trajectories were developed for different future mortality levels relative to the status quo. Survival of the pelagic immature ages was not varied. Survival of benthic immature and adult components was varied by rescaling the 1995 estimates of instantaneous mortality by different multipliers (+/- 10% and +/- 20%):

Change in instantaneous mortality rate

	Proportionate Survival Rate				
Scenario	-20%	-10%	Status Quo	+10%	+20%
Pelagic	.44	.44	.44	.44	.44
Benthic immature	.80	.77	.75	.73	.71
Adult	.91	.90	.89	.88	.87

Under status quo mortality rates, the target level of 10,000 nesting females per year is obtained in 2015 (Figure F). As of 2010, only 6100 females would be expected to nest. By reducing instantaneous mortality on benthic immature and adult turtles by 10%, 10,000 nesting females are obtained in 2012. A 20% reduction in instantaneous mortality would be required to obtain 10,000 nesting females by 2010.

Recovery times would be eroded by any increase in mortality of benthic immatures and adults. A 10% increase in instantaneous mortality would delay projected recovery to 2020; a 20% increase would delay recovery until 2026. If the 33% decrease in instantaneous mortality rate estimated for the 1990-1995 period were lost after 1995, 10,000 nesters would not be obtained until 2046.

Uncertainty

These analyses include no quantification of uncertainty associated with estimates of 1995 standing stock size and mortality rates, nor with projected hatchling production. The assumptions required to implement the model were so extensive that uncertainty around estimates could not be quantified.

The model works to minimize the annual differences between observed nests and predicted nests. Both components involve potentially significant sources of error. In the case of observed nests, we felt that although it could have been possible that the nesting range for Kemp's ridleys may have extended beyond Rancho Nuevo before the establishment of additional camps in 1989-1990, no reliable estimates were available for that portion of the range in the early part of the time series. Thus, estimates of nests before 1989 would be biased low to an unknown degree. In the case of predicted nests, the quantitative relationship between population size and number of nests produced annually is imprecisely defined. The relationship depends on remigration rate, nests per female, age at sexual maturity and sex ratio. Each of these components has associated but currently undefined variance. An approximate coefficient of variation for estimates of population size (from nests) which incorporated only variance in remigration rate and number of nests per female is about 33%, based on Monte Carlo simulations incorporating potential patterns in variation.

Model results are highly sensitive to benthic immature survival rate and age at sexual maturity, based on results of initial sensitivity analyses. The benthic immature survival rate shows variation not only in terms of the variance of the estimate itself, but also as function of the area and size range serving as the basis for an estimate. Animals likely mature sexually over a wider age range than ranges evaluated here, but the extent of that range and the shape of the maturity schedule is unknown.

The model formulation contains several structural features which may contribute to unreliable results. The mortality rate of adults is determined at least in part by the rate of decline of nesting females between 1966 and 1978. The mortality rate of pelagic juveniles is critically dependent on mortality rates of benthic immature and adult stages, and parameter estimates could be highly correlated. This correlation is as yet unevaluated.

Projection results are intended for descriptive rather than predictive purposes. The time horizon for these projections is long, given the short time scale for potential changes in anthropogenic mortality. Assumption of stable population dynamics processes over 20 years is unrealistic, especially when large changes in anthropogenically or environmentally induced mortality could occur rapidly and unpredictably. Moreover, the likelihood of obtaining trajectories similar to the ones presented here is liable to be small: a reasonable range of trajectory outcomes depends on uncertainty in starting stock size and parameter estimates.

Analysis of Trends - Strandings, Nests, and Hatchling Production

It may be difficult to distinguish trends in the number of stranded turtles from trends in population size (Crowder et al., 1995 (*SC strandings analysis*)). While the number of Kemp's ridley strandings increased in 1993, 1994 and 1995, the population may also have increased, given the large number of hatchlings released from Rancho Nuevo in preceding years and possible decreases in mortality due to TED implementation. We compared these trends by calculating the slopes of log-transformed strandings, hatchling production 2 years prior to each year's strandings (assumes a pelagic phase of 2 years), and Rancho Nuevo nest data (Table 1, Figure 1). We used the strandings data for all U.S. regions, 1986 - 1995, divided into the following categories: small benthic immatures (20 - 30 cm SCL), representing the youngest age class(es) present in benthic feeding habitats, benthic immatures (20 - 60 cm SCL), adults (> 60 cm SCL), and all benthic turtles (20+ cm SCL), which included all unmeasured animals. These four categories represented most of the turtles in the strandings record; pre-benthic (post-hatchling) sized turtles occasionally strand, but were excluded from this analysis. In general, strandings were far more variable from year to year than nests or hatchling production. The slopes, which provided the rate of increase in proportion per year for each data set, were highly dependent on which years were used. The TEWG has proposed that strandings and population trends be calculated over a 5-year interval for setting Interim Strandings Limits (ISL). The regression analysis indicates that from 1991 to 1995, benthic immature strandings and total benthic turtle strandings increased far more rapidly than population size, as indexed by nests or hatchling production. Adult strandings did not show a significant slope over this (or any other) time period; very few adult Kemp's ridleys strand in any one year, and this regression analysis was unable to detect a trend. The 1990 to 1995 analysis shows a much lower increase in benthic immature and total benthic turtle strandings, but the slopes were no longer significant at the $p < 0.05$ level. Over the entire data set (1986 - 1995), strandings have no significant slope. Analyzing the trend over this time period does not account for changes in mortality due to TED regulations; if TEDs are affecting mortality in the benthic immature and adult stages, we would anticipate a change in the strandings trend some time around 1990. From 1986 - 1990, none of the data measured showed a significant trend.

Strandings should not increase at a faster rate than population size. Further analysis of the residuals from this analysis may be warranted, as well as a statistical analysis of differences among the slopes; however, the low sample size may be prohibitive. Strandings in 1996 may help determine if the low strandings in 1991 and 1992 or the high strandings in 1994 and 1995 are more typical under current management.

Interim Stranding Limit (ISL)

We concluded that estimation of the maximum number of individual Kemp's ridley sea turtles that can be taken incidentally to commercial fishing activities without preventing the recovery of the species cannot be made at this time. Estimates of total population size and current mortality rates were found to be too sensitive to current data deficiencies and estimation model structure to be reliable. Moreover, any apportioning of those mortality rates to commercial fishing activities cannot be made from available data.

We also concluded that the best proxy for estimation of take at this point should be developed from strandings data and other direct empirical observations of stock status. For this species, those empirical observations presently consist of hatchling production and nest levels. Strandings represent an imperfect but directly observable quantity, although quantitative relationships between strandings levels, stock size, and mortality rates (including their fishery and natural components) remain undeveloped. We anticipate that these relationships will be developed in the future, and consequently only recommend use of the proposed proxy approach for the upcoming year.

We identified an Interim Stranding Limit (ISL) based on strandings during the recent period of population growth assuming these strandings are sustainable in the short term. We suggest that the species will continue to recover at a reasonable rate if stranding levels don't exceed numbers that are indexed to increases in nesting population (as a proxy for total population) in a conservative risk-averse manner. As a precautionary measure, stranding levels should not increase faster than one-half the observed rate of increase in nests. Observed rate of increase in nests is estimated as the slope of the natural log of nest levels over the past 3 and 5 years, transformed to a linear (proportionate) scale. The annual percentage increase is then divided by two. The resulting percentage increase is applied to recent stranding levels, as estimated by (the mean + one standard deviation above the mean) of stranding levels over the past 3 and 5 years. This approach contains substantial risk of not being responsive to changes in mortality on juveniles (especially at early ages) until those individuals have recruited to the nesting population. Additionally, the extent to which this approach is risk-averse has not been evaluated (e.g., in the manner in which the performance of Potential Biological Removals (PBR) has been evaluated for marine mammals). This further supports the use of this proxy for a single year only. (proxy runs Table 5)

CONCLUSIONS

1. The Kemp's ridley population appears to be in the early stage of exponential expansion. Over the period 1986 to 1995, the rate of increase in the annual number of nests accelerated. This upward trend should continue with increased hatchling production and protection at sea with the use of turtle excluder devices.

2. Significantly increased strandings occurred throughout the southeast and in particular in Texas and Louisiana in 1994 and 1995. Technical and intentional violations in TED gear installation, intense pulse fishing and use of less effective TEDs were all implicated as causes of the increased mortality. It is also likely that an increase in the number of benthic immature Kemp's ridley turtles resulting from increased hatchling production and the use of TEDs may be providing an increased pool of turtles available for all sources of mortality including bottom trawls and gill nets.

3. Adults declined from a population that produced an estimated 6,000 nests in 1966 to a population that produced 924 nests in 1978 and a low of 702 nests in 1985. In 1995, the number of nests increased to 1,930.

4. The estimation of the maximum number of individual Kemp's ridley sea turtles that can be taken incidentally to commercial fishing activities without preventing the recovery of the species cannot be made at this time because of data deficiencies. Strandings represent an imperfect but directly observable quantity, although relationships between stranding levels, stock size, and mortality rates remain unknown. An Interim Stranding Limit (ISL) was developed using 3 and 5 year running averages of strandings which should not increase annually at a rate greater than the population rate of increase. A risk averse approach includes using one half of the estimated annual rate of increase in nests to calculate the ISL. The extent to which this approach is risk averse needs to be evaluated, particularly relative to benthic immature mortality. This approach should only be used as a proxy for one year.

5. Modeling efforts suggest that survival rates of benthic immatures and adults must be reduced post 1990 to balance egg inputs and number of nesting females. This supports the idea of a TED effect, but we have not yet documented such an effect empirically.

6. It is currently projected that this species could reach the intermediate recovery goal of 10,000 nesting females by the year 2020 given that the assumptions of age to sexual maturity and age specific survival rates are correct. The extent that these assumptions are violated will determine if this intermediate goal is achieved. However, if the current annual rate of increase in nests continues, this goal can be achieved. Note that much information is missing regarding these projections and sensitivity analysis give very different results with different values.

7. The increase in hatchling production at Rancho Nuevo in recent years is the result of improvements in turtle camp operations, which produced higher rates of collection of eggs from a still small but increasing number of nesting females, increased protection of those eggs, and

increased hatch rates. If the annual number of nesters continues to increase, and the nesting area continues to expand, with a consequent increase in number of nests, it will take even greater efforts at the turtle camps to assure that these positive indications of initial recovery result in increased hatchling production.

8. Continued, and perhaps expanded, human intervention through protection of the Rancho Nuevo nesting beach and other nesting beaches is necessary for recovery during the foreseeable future, otherwise the gains that have been made could be lost. We are concerned about human encroachment, access and potential development that could adversely impact the Rancho Nuevo nesting beach, especially the planned expansion of fishing facilities and extension of the Gulf Intracoastal Waterway from the U.S. to Tampico, Mexico (USFWS and NMFS 1992).

9. The status of this species and ISL needs to be evaluated annually.

RECOMMENDATIONS

Many of the following recommendations will produce data necessary to improve the parameters of population and assessment models being applied to sea turtles. Reduction of the level of uncertainty in these parameters will significantly increase the value of models and the accuracy of model results.

1. Long-term, in-water indices of abundance are needed to identify relative abundance of sea turtles, particularly Kemp's ridleys, over time and to detect changes in size composition with implications regarding recruitment. Studies should be conducted in coastal waters along the west coast of Florida, in Louisiana, in Texas, in Pamlico Sound in North Carolina, in Georgia waters, and in one northern embayment. All turtles caught should be tagged with PIT tags as well as flipper tags, and should be scanned for existing PIT tags. Standardized catch per unit effort (CPUE) methods should be employed to provide site-to-site and long-term comparability. Projects should be consistent and long-term, with data collected in a statistically defensible manner. Habitat use and tracking studies conducted as adjuncts to abundance index studies should not compromise the effectiveness of the primary goal of abundance indexing. Projects which provide an assessment of trends in juvenile abundance should be given priority, to provide information on trends that will not be reflected immediately on nesting beaches.

2. Information is needed regarding the location of mortality of stranded sea turtles to infer possible causes of strandings. Oceanographic conditions affecting stranding rates (considering decomposition rates and oceanic scavenging) should be examined to determine sea turtle carcass landfall patterns and to identify conditions or areas where carcasses would not be likely to strand.

3. Stranding network index areas within the southeast region should be developed for long-term standardized coverage by the STSSN. The statistical comparability of the index areas should be primary considerations in the identification and monitoring of the study sites.

4. Development of standardized PIT tag techniques and equipment should be developed. A pit tag registry should be initiated and maintained.

5. Kemp's ridley nesting females should be intercepted on the beach at levels approaching 100%. Saturation tagging should be conducted, with PIT tags. All females should be scanned for previously placed PIT tags.

6. PIT tag scanners should be issued to representative STSSN participants in each state to maximize the mark-recapture components of long-term sea turtle studies.

7. Efforts to expand the shrimp vessel observer program should continue. Scientists should work with industry to assess sea turtle catch per unit effort in various areas, including inshore embayments, nearshore waters and offshore waters to determine whether there are some areas and seasons in which TED regulations need to be modified.

8. Age and growth studies should be conducted for Kemp's ridleys in both the Gulf and Atlantic.

9. Physiological studies to determine the effects of forced and repeated submersion should be continued.

10. Aerial surveys of shrimp vessels should be conducted to provide resolution on the distribution of fishing effort.

11. The utility of population models could be increased through improved age-class survival estimates for ridleys and loggerheads. Life history data collection from nesting beach activities, stranding monitoring, and in-water abundance surveys should support this research goal. More data on sex ratios, maturity, breeding condition and accurate sizes can be gleaned from examination of stranded carcasses. Laparoscopic studies could yield the percent of neophytes in a nesting population and more saturation tagging is needed to assess female survivorship and remigration.

12. Shrimp fishing effort reduction measures, including closures, gear restrictions, or other measures should be evaluated to further reduce nearshore shrimping effort for increased sea turtle protection, while remaining consistent with state management goals and local shrimping industry interests.

13. Other sources of fisheries-related mortality that have high nearshore effects, such as gillnets, crab trawling, long lines, purse seines, recreational hook and line, and bottom trawl fisheries need to be evaluated.

14. Additional assistance should be provided to Mexico in conducting a thorough statistical analysis of the Kemp's ridley nesting beach data. This should include counts of nests, eggs, hatchlings and nesters, mark-recapture data, size distributions, and adult female recruitment.

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Table 1. Time series of nests and hatchlings numbers used in these analyses (1995 actual hatchlings were 120,038)

Year	Nests	Hatchlings Released
1966	6000	29100
1967	5500	24100
1968	5200	15000
1969	4000	28400
1970	3000	31400
1971	2000	13100
1972	1800	14600
1973	1600	23500
1974	1400	23500
1975	1200	11100
1976	1100	36100
1977	1050	30100
1978	924	48009
1979	954	63996
1980	868	37378
1981	897	53282
1982	750	48007
1983	746	32921
1984	798	58124
1985	702	51033
1986	744	48818
1987	737	44634
1988	842	62218
1989	878	66752
1990	992	74339
1991	1155	76238
1992	1275	92116
1993	1184	84210
1994	1566	107687
1995	1930	126000

Table 2. Sensitivity analysis of the 10 year to maturity age-based model for Kemp's ridley sea turtles. Benthic immature mortality rate was fixed for each run; adult mortality, pelagic immature mortality and the post-1990 decrease in mortality rates were chosen for a least-squares fit of predicted nests to observed nests for 1978 - 1995. Annual cohort size defined by actual number of hatchlings released from corrals 1966 - 1995. Reproduction parameters fixed for all runs: nests/female = 2.5, remigration interval = 2 years. Sex ratio = 1:1.

benthic immature (age 2 - 9)		pelagic immature (age 0 - 1)		adult(age 10+)		post-1990 multipler for benthic imm. and adult z	sum of squares
z	s	z	s	z	s		
0.36	0.70	1.06	0.35	0.17	0.84	0.64	60801
0.4	0.67	0.90	0.41	0.17	0.84	0.66	59099
0.44	0.64	0.74	0.48	0.17	0.84	0.69	57667
0.48	0.62	0.58	0.56	0.17	0.84	0.70	56450
0.52	0.59	0.41	0.66	0.17	0.84	0.72	55407
0.56	0.57	0.25	0.78	0.17	0.84	0.74	54506

Table 3. Estimated year that the population will reach 10,000 nesting females per season, assuming that current conditions (survival and growth rates) remain constant, for models with and without density-dependent limitation in nest survival (see text for details), and the post-1990 decrease in instantaneous mortality rates necessary to reach the recovery goal of 10,000 nesting females by 2020.

	Year to reach 10,000 nesters		post-1990 change in mortality needed to reach 10,000 nesters by the year 2020	
benthic immature z	no density-dependence	with density-dependence	no density-dependence	with density-dependence
0.36	2014	2015	0.76	0.72
0.4	2014	2015	0.79	0.74
0.44	2014	2015	0.8	0.76
0.48	2013	2015	0.81	0.78
0.52	2013	2014	0.83	0.8
0.56	2013	2014	0.84	0.81

Table 4. Population size estimates from the sensitivity analysis. Assume 1:1 sex ratio.

	population size in 1995		
benthic immature z	adults	benthic immature	pelagic immature
0.36	2,963	36,902	163,360
0.4	2,968	47,603	169,889
0.44	2,973	61,590	177,554
0.48	2,977	79,936	186,553
0.52	2,981	104,075	197,118
0.56	2,984	135,927	209,520

Table 5. 3 and 5 Year Running Averages and Interim Stranding Limit.

Five Year Average Model

	Nests Multiplier Of Increase	Stranding Mean + 1 Std Dev		Mean	Total ISL +1 Std DevStrandings	Observed			
		0.5							
1986-90	1.0779	0.0390	250.40	314.23					
1991	260.16	326.47		186					
1987-91	1.1121	0.0560		221.80	272.24	1992	234.23	287.50	168
1988-92	1.1167	0.0584		213.00	269.10	1993	225.43	284.80	362
1989-93	1.0886	0.0443		241.80	329.29	1994	252.51	343.88	578
1990-94	1.0978	0.0489		320.40	485.88	1995	336.06	509.63	521
1991-95	1.1309	0.0654		363.00	550.45	1996	386.75	586.46	

Three Year Average Model

Period		Nests Multiplier Of Increase	Strandings Mean	+ 1 Std Dev	Total ISL	Mean	Observed +1 Std DevStrandings	
1986-1988	1.0638	0.0319	253.00	318.89	1989	261.07	329.06	185
1987-1989	1.0915	0.0457	205.00	222.58	1990	214.38	232.76	308
1988-1990	1.0854	0.0427	237.00	300.66	1991	247.12	313.51	186
1989-1991	1.1469	0.0735	226.33	297.06	1992	242.96	318.89	168
1990-1992	1.1337	0.0669	220.67	296.83	1993	235.42	316.68	362
1991-1993	1.0125	0.0062	238.67	345.85	1994	240.16	348.01	578
1992-1994	1.1068	0.0534	369.33	574.43	1995	389.06	605.12	521
1993-1995	1.2767	0.1384	487.00	598.94	1996	554.39	681.82	

Figure 1.

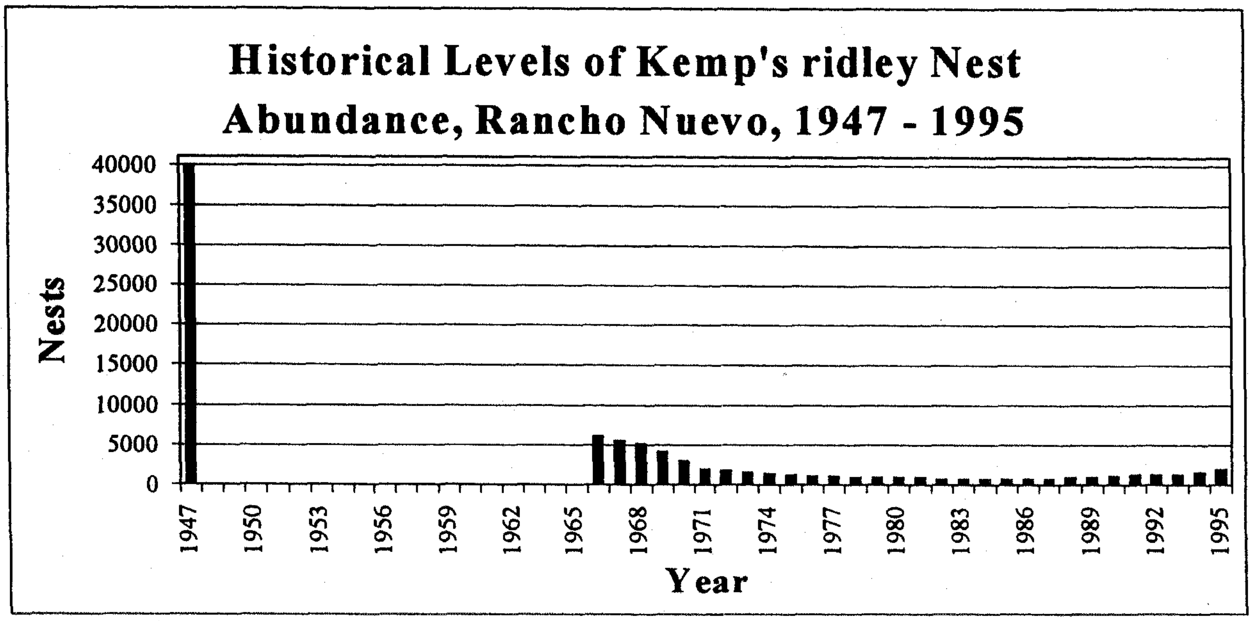
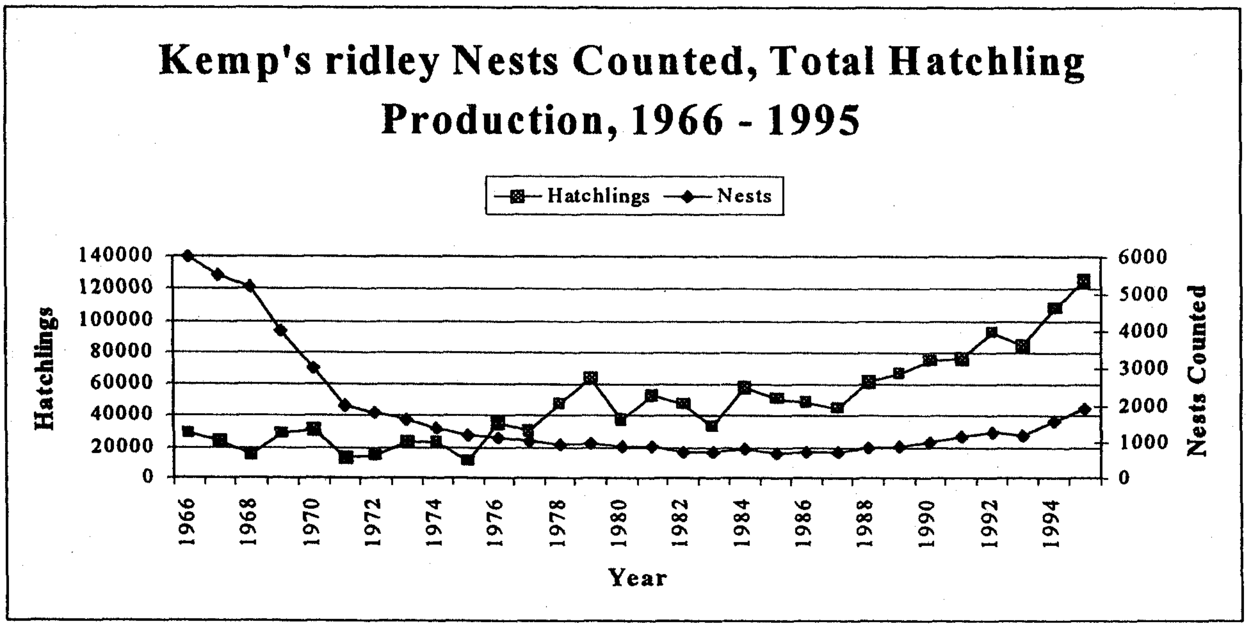


Figure 2.



Appendix 1

	Age	1966-1990 Proportionate Survival	Instantaneous Mortality	post-1990 Proportionate Survival	Instantaneous Mortality
Hatchling	0 - 1	0.442	0.817	0.442	0.817
Benthic Immature	2 - 9	0.657	0.42	0.753	0.283
Adult	10+	0.841	0.173	0.89	0.117

post-1990 mortality multiplier 0.675

Model Fit

Year	Observed nests	Predicted nests	Residual
1966	6000	6000.00	0.00
1967	5500	5045.10	454.90
1968	5200	4242.17	957.83
1969	4000	3567.03	432.97
1970	3000	2999.33	0.67
1971	2000	2521.99	-521.99
1972	1800	2120.61	-320.61
1973	1600	1783.12	-183.12
1974	1400	1499.33	-99.33
1975	1200	1260.71	-60.71
1976	1100	1183.34	-83.34
1977	1050	1097.10	-47.10
1978	924	986.03	-62.03
1979	954	949.41	4.59
1980	868	931.32	-63.32
1981	897	838.59	58.41
1982	750	766.97	-16.97
1983	746	744.45	1.55
1984	798	725.52	72.48
1985	702	657.07	44.93
1986	744	705.42	38.58
1987	737	720.65	16.35
1988	842	809.33	32.67
1989	878	951.61	-73.61
1990	992	958.49	33.51
1991	1155	1111.42	43.58
1992	1275	1255.97	19.03
1993	1184	1327.41	-143.41
1994	1562	1606.09	-44.09
1995	1930	1856.77	73.23

Residual
Sum-of-Squares, 1978-1995 58352.72

Figure A1.1 Observed and model predicted Kemp's ridley nests, 1978-1995

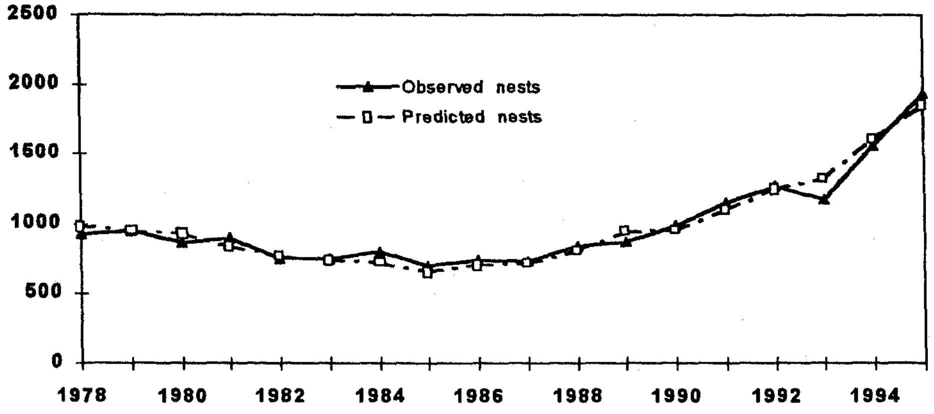


Figure A1.2 Residuals: Observed - Expected Nests, 1978-1995

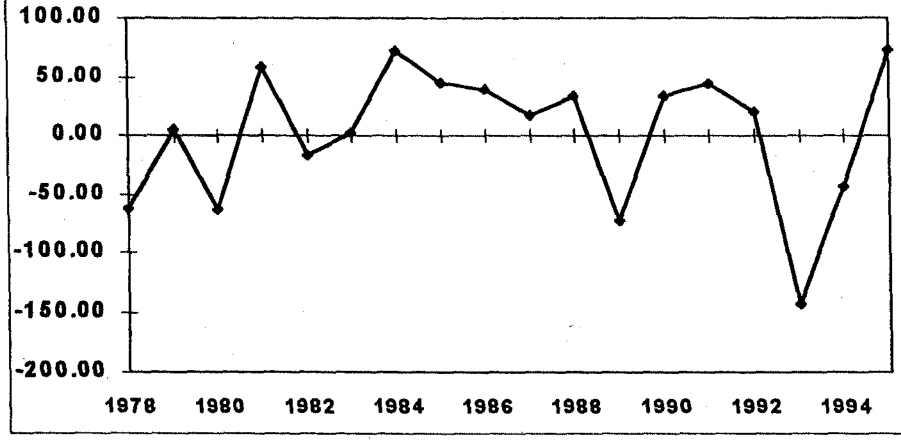


Figure A1.3 Model estimated adult and benthic immature abundance, 1966-1995

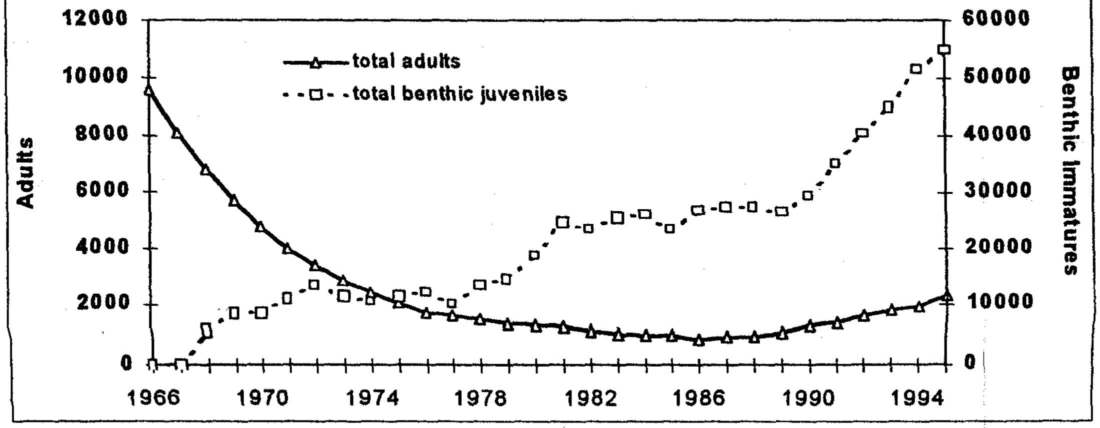


Figure A1.4 Model estimated benthic immature deaths and observed strandings, 1986-1995

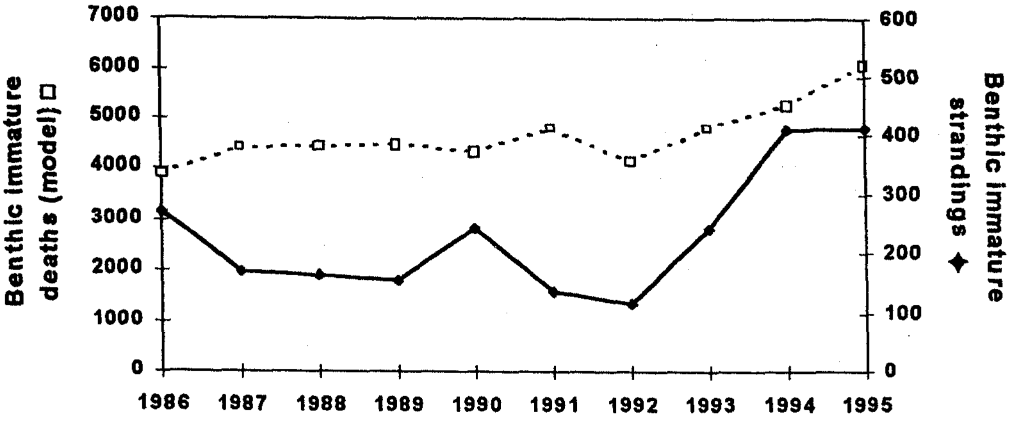


Figure A1.5 Model estimated proportion of neophyte nesting females, 1966-1995

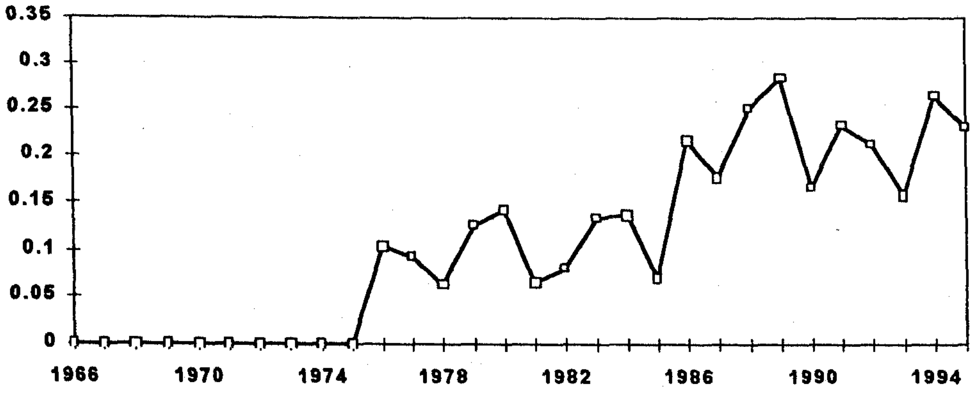
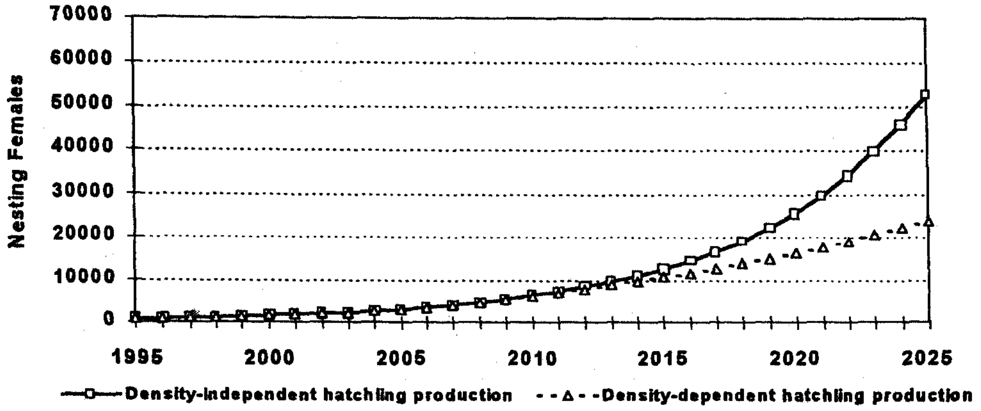


Figure A1.6 Model projections - estimated number of nesting females with and without density-dependent hatchling production



Appendix 1

Estimated
Female Standing Stock in 1995

Age	Number
0	63000.00
1	23783.97
2	8215.54
3	6769.01
4	4219.68
5	3099.15
6	2096.08
7	1471.56
8	693.62
9	498.46
10	342.37
11+	1143.04

Appendix 2

	Age	1966-1990 Proportionate Survival	Instantaneous Mortality	post-1990 Proportionate Survival	Instantaneous Mortality
Hatchling	0 - 1	0.46	0.776	0.46	0.776
Benthic Immature	2 - 9	0.657	0.42	0.746	0.293
Adult	10+	0.826	0.192	0.875	0.134

post-1990 mortality multiplier 0.697

Model Fit

Year	Observed nests	Predicted nests	Residual
1966	6000	7500.00	-1500.00
1967	5500	6192.13	-692.13
1968	5200	5112.33	87.67
1969	4000	4220.83	-220.83
1970	3000	3484.79	-484.79
1971	2000	2877.10	-877.10
1972	1800	2375.39	-575.39
1973	1600	1961.16	-361.16
1974	1400	1619.17	-219.17
1975	1200	1336.81	-136.81
1976	1100	1237.39	-137.39
1977	1050	1132.34	-82.34
1978	924	1003.79	-79.79
1979	954	959.23	-5.23
1980	868	936.22	-68.22
1981	897	833.14	63.86
1982	750	754.93	-4.93
1983	746	731.25	14.75
1984	798	711.70	86.30
1985	702	638.59	63.41
1986	744	693.09	50.91
1987	737	710.52	26.48
1988	842	807.18	34.82
1989	878	960.44	-82.44
1990	992	964.69	27.31
1991	1155	1122.13	32.87
1992	1275	1266.38	8.62
1993	1184	1329.69	-145.69
1994	1562	1607.85	-45.85
1995	1930	1850.01	79.99

Residual
Sum-of-Squares, 1978-1995 69763.60

Appendix 2

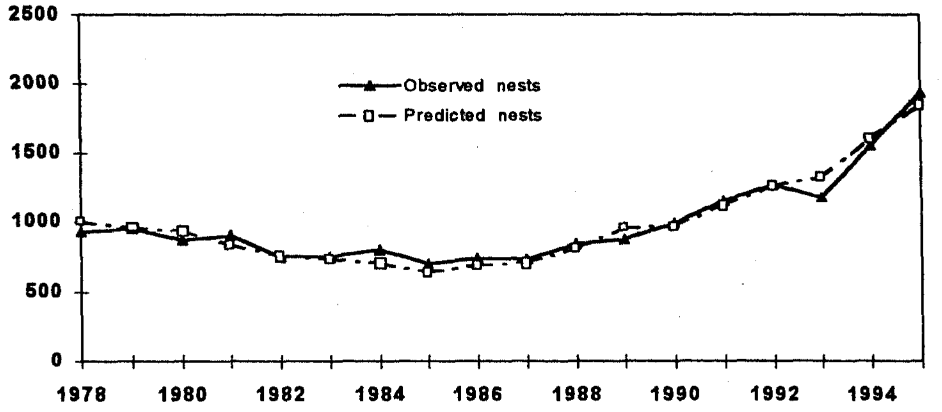


Figure A2.1 Observed and model predicted Kemp's ridley nests, 1978-1995

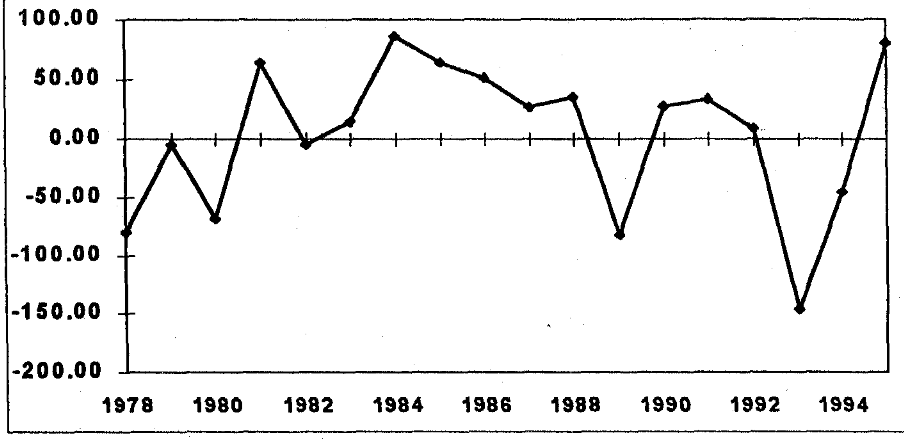


Figure A2.2 Residuals: Observed - Expected Nests, 1978-1995

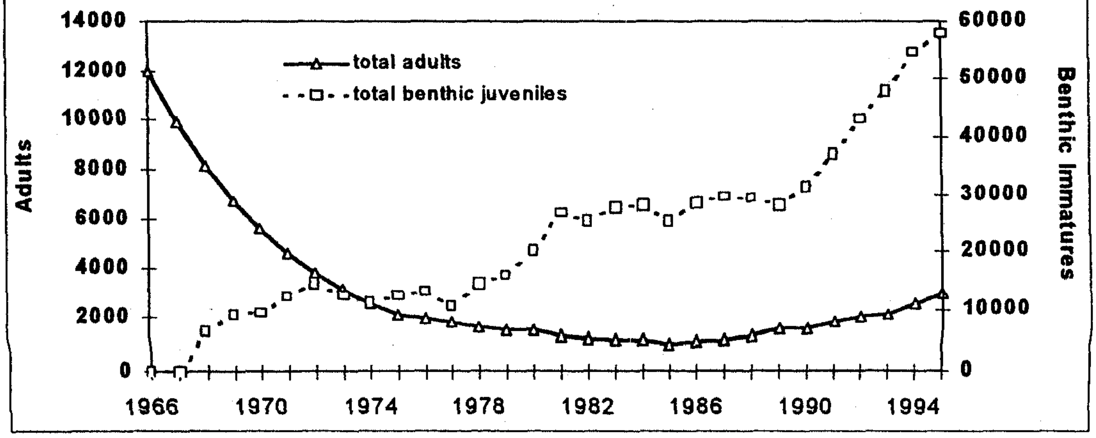


Figure A2.3 Model estimated adult and benthic immature abundance, 1966-1995

Figure A2.4 Model estimated benthic immature deaths and observed strandings, 1986-1995

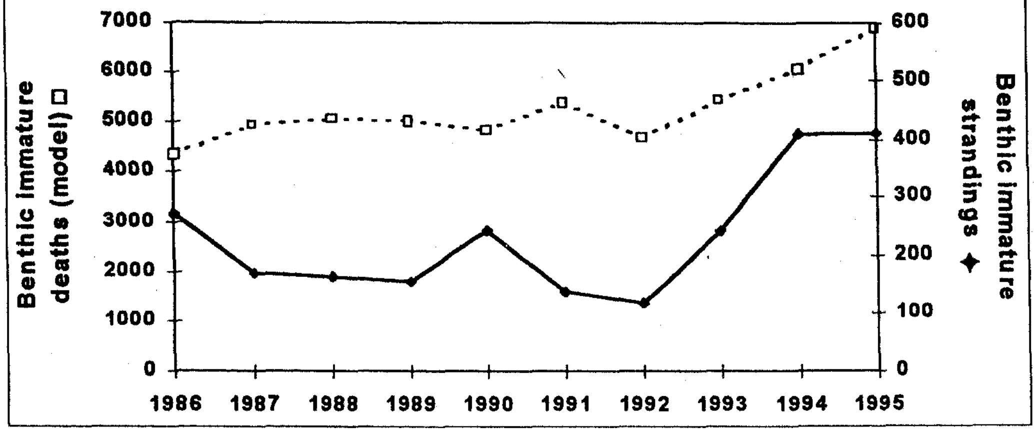


Figure A2.5 Model estimated proportion of neophyte nesting females, 1966-1995

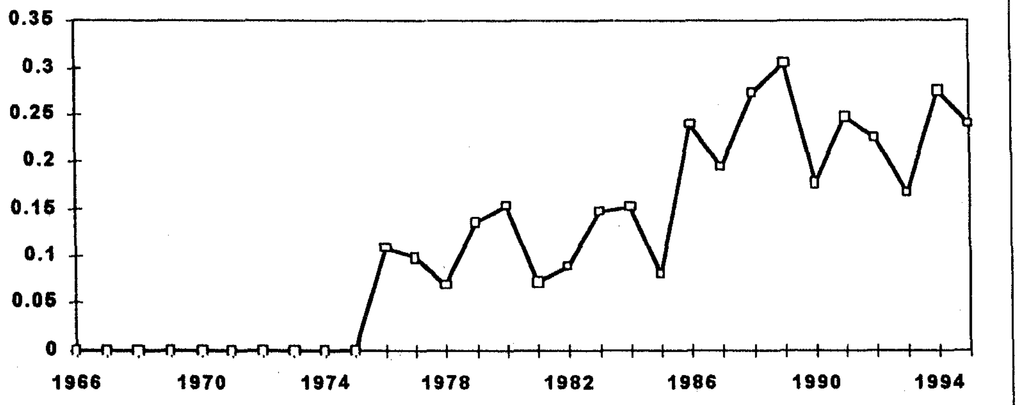
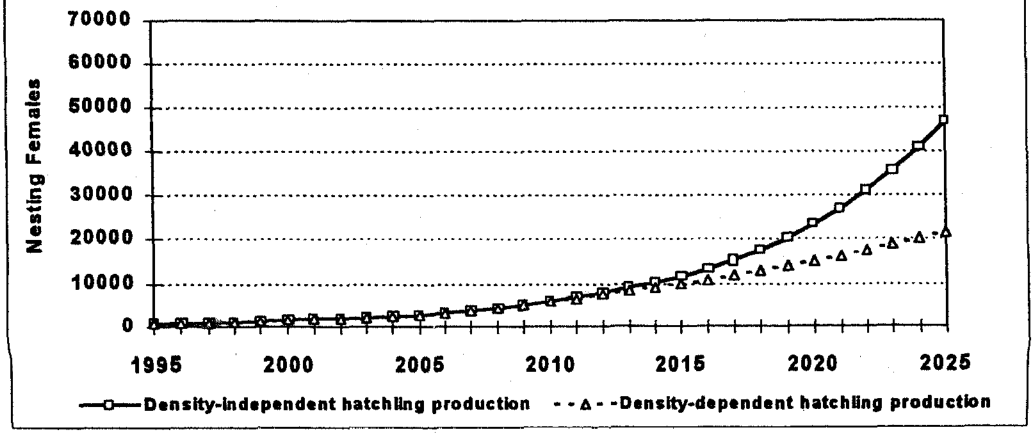


Figure A2.6 Model projections - estimated number of nesting females with and without density-dependent hatchling production



Appendix 2

Estimated
Female Standing Stock in 1995

Age	Number
0	63000.00
1	24769.64
2	8910.59
3	7273.91
4	4492.57
5	3269.12
6	2190.63
7	1523.74
8	718.22
9	516.14
10	354.51
11+	1125.49

Appendix 3

		1966-1990		post-1990	
	Age	Proportionate Survival	Instantaneous Mortality	Proportionate Survival	Instantaneous Mortality
Hatchling	0 - 1	0.416	0.876	0.416	0.876
Benthic Immature	2 - 9	0.657	0.42	0.764	0.269
Adult	10+	0.861	0.15	0.909	0.096

post-1990 mortality multiplier 0.64

Model Fit

Year	Observed nests	Predicted nests	Residual
1966	6000	4500.00	1500.00
1967	5500	3873.80	1626.20
1968	5200	3334.74	1865.26
1969	4000	2870.69	1129.31
1970	3000	2471.22	528.78
1971	2000	2127.34	-127.34
1972	1800	1831.31	-31.31
1973	1600	1576.47	23.53
1974	1400	1357.10	42.90
1975	1200	1168.25	31.75
1976	1100	1115.22	-15.22
1977	1050	1050.74	-0.74
1978	924	960.99	-36.99
1979	954	934.16	19.84
1980	868	922.36	-54.36
1981	897	843.32	53.68
1982	750	780.93	-30.93
1983	746	760.71	-14.71
1984	798	743.31	54.69
1985	702	681.66	20.34
1986	744	722.69	21.31
1987	737	735.42	1.58
1988	842	813.80	28.20
1989	878	941.44	-63.44
1990	992	951.13	40.87
1991	1155	1097.53	57.47
1992	1275	1241.80	33.20
1993	1184	1323.42	-139.42
1994	1562	1603.26	-41.26
1995	1930	1866.13	63.87

Residual
Sum-of-Squares, 1978-1995 48750.37

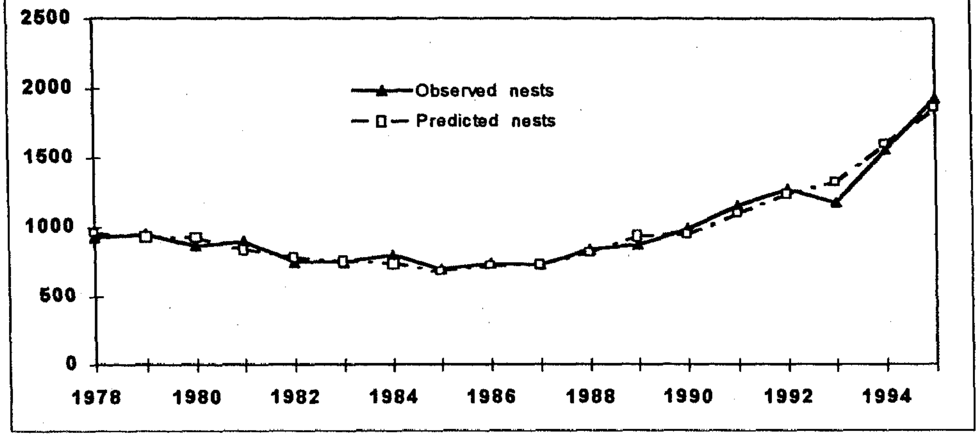


Figure A3.1 Observed and model predicted Kemp's ridley nests, 1978-1995

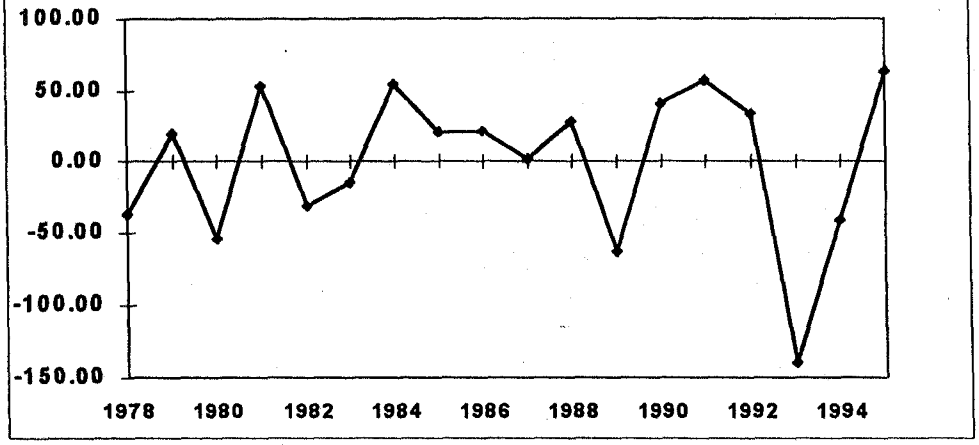


Figure A3.2 Residuals: Observed - Expected Nests, 1978-1995

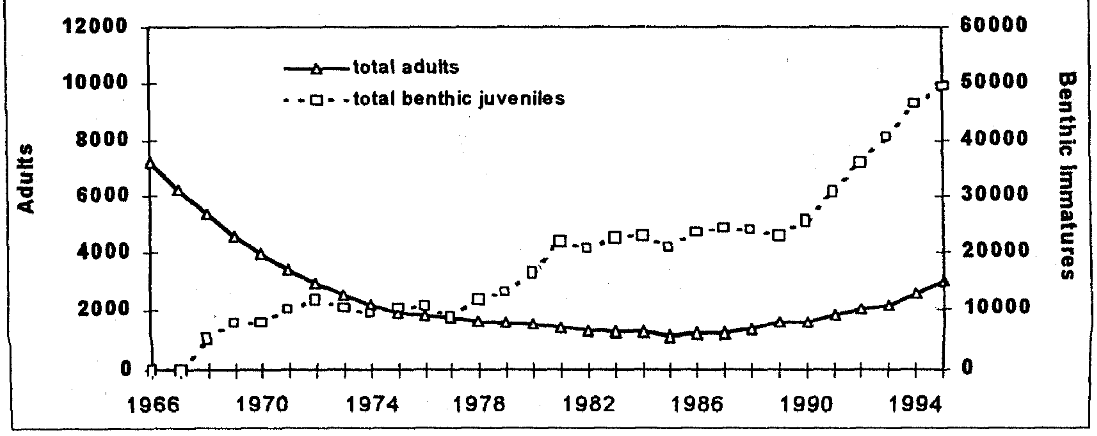


Figure A3.3 Model estimated adult and benthic immature abundance, 1966-1995

Figure A3.4 Model estimated benthic immature deaths and observed strandings, 1986-1995

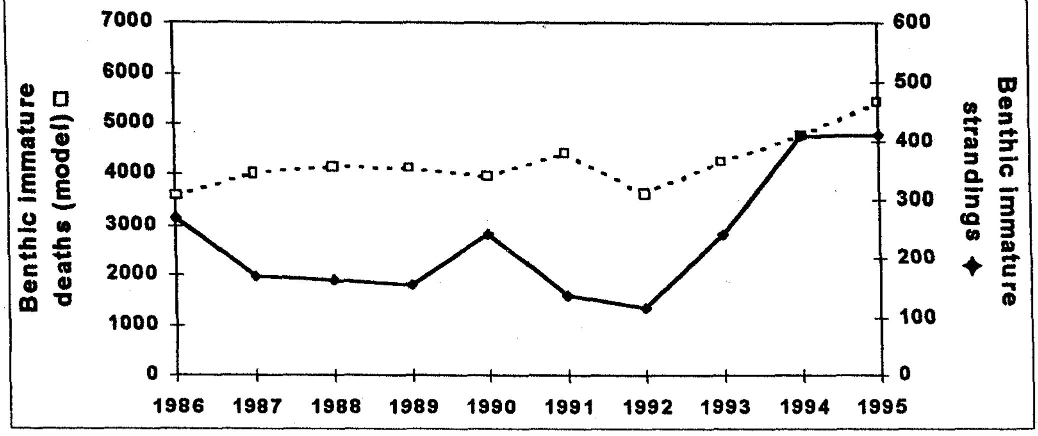


Figure A3.5 Model estimated proportion of neophyte nesting females, 1966-1995

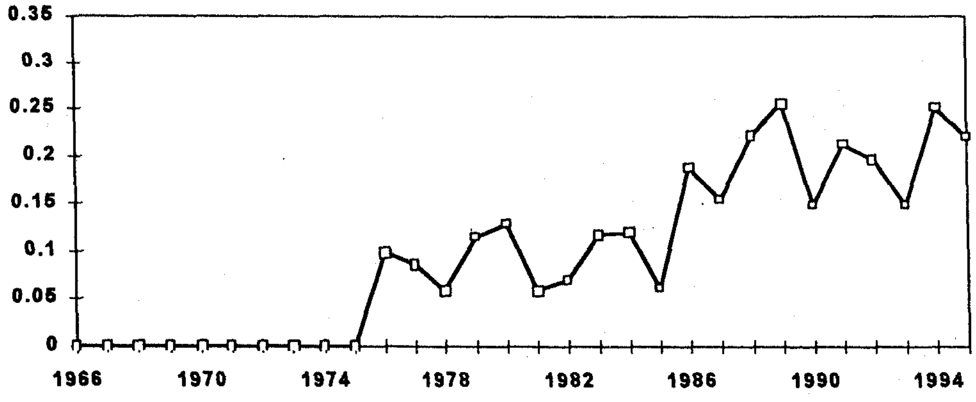
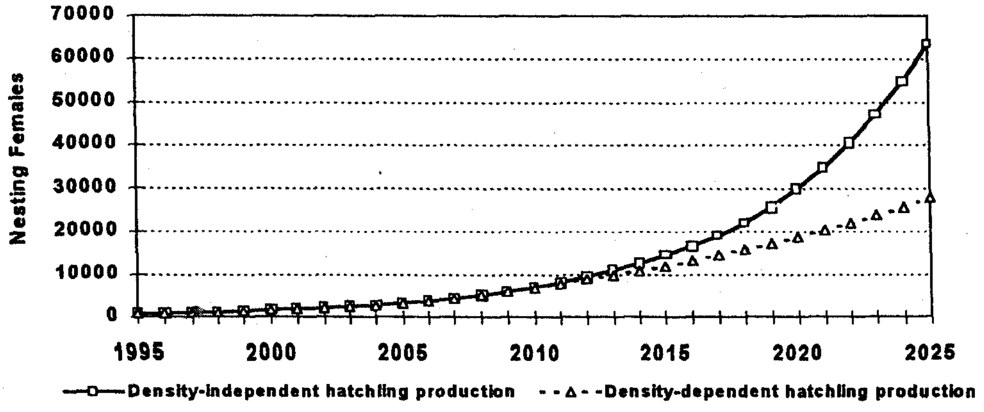


Figure A3.6 Model projections - estimated number of nesting females with and without density-dependent hatchling production



Estimated
Female Standing Stock in 1995

Age	Number
0	63000.00
1	22420.24
2	7300.41
3	6104.38
4	3861.89
5	2878.51
6	1975.78
7	1407.71
8	663.53
9	476.84
10	327.52
11+	1165.38

Appendix 4

		1966-1990		post-1990	
	Age	Proportionate Survival	Instantaneous Mortality	Proportionate Survival	Instantaneous Mortality
Hatchling	0 - 1	0.714	0.337	0.714	0.337
Benthic Immature	2 - 11	0.657	0.42	0.738	0.304
Adult	12+	0.847	0.166	0.887	0.12

post-1990 mortality multiplier 0.723

Model Fit

Year	Observed nests	Predicted nests	Residual
1966	6000	6000.00	0.00
1967	5500	5083.29	416.71
1968	5200	4306.65	893.35
1969	4000	3648.66	351.34
1970	3000	3091.20	-91.20
1971	2000	2618.91	-618.91
1972	1800	2218.78	-418.78
1973	1600	1879.79	-279.79
1974	1400	1592.59	-192.59
1975	1200	1349.26	-149.26
1976	1100	1143.12	-43.12
1977	1050	968.47	81.53
1978	924	959.43	-35.43
1979	954	927.91	26.09
1980	868	857.75	10.25
1981	897	862.29	34.71
1982	750	880.46	-130.46
1983	746	808.49	-62.49
1984	798	754.67	43.33
1985	702	751.56	-49.56
1986	744	748.93	-4.93
1987	737	687.50	49.50
1988	842	754.82	87.18
1989	878	783.20	94.80
1990	992	892.75	99.25
1991	1155	1135.01	19.99
1992	1275	1231.84	43.16
1993	1184	1453.04	-269.04
1994	1562	1653.56	-91.56
1995	1930	1747.59	182.41

Residual
Sum-of-Squares, 1978-1995 173719.43

Figure A4.1 Observed and model predicted Kemp's ridley nests, 1978-1995

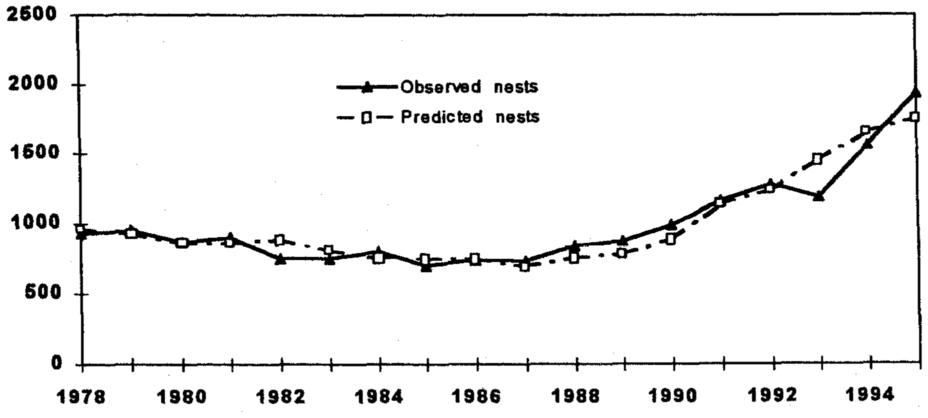


Figure A4.2 Residuals: Observed - Expected Nests, 1978-1995

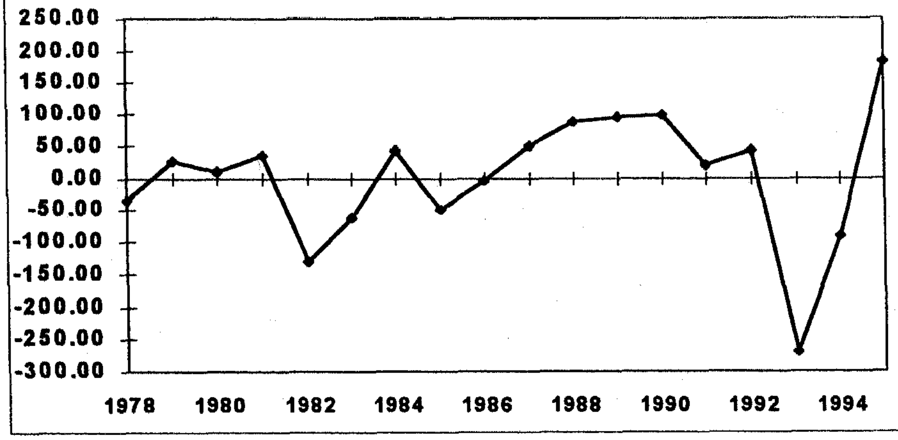
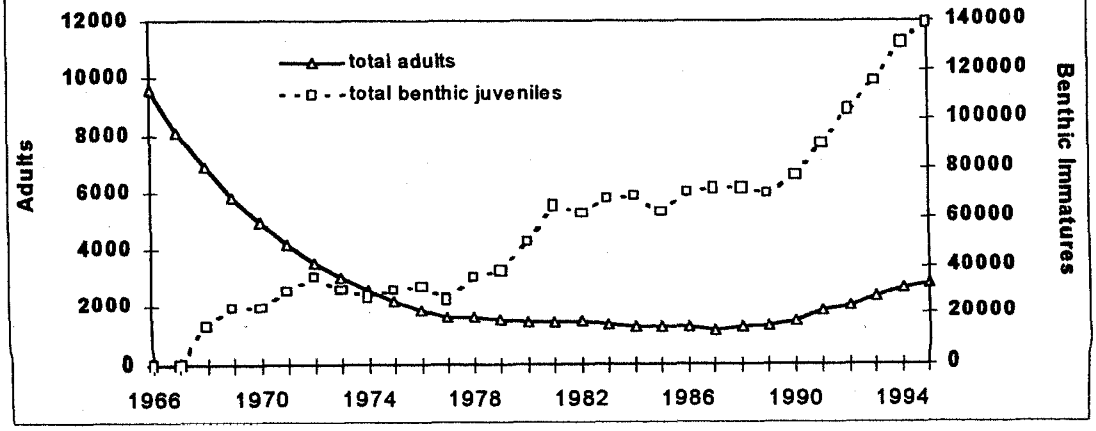


Figure A4.3 Model estimated adult and benthic immature abundance, 1966-1995



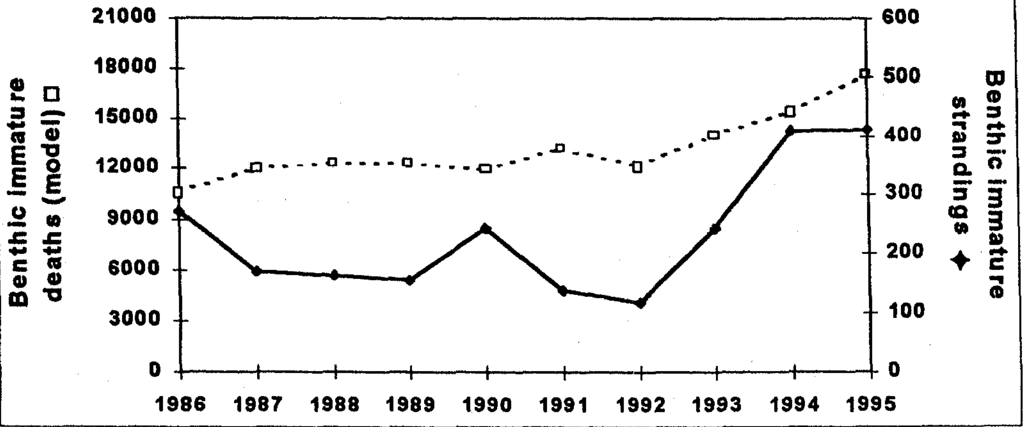


Figure A4.4 Model estimated benthic immature deaths and observed strandings, 1986-1995

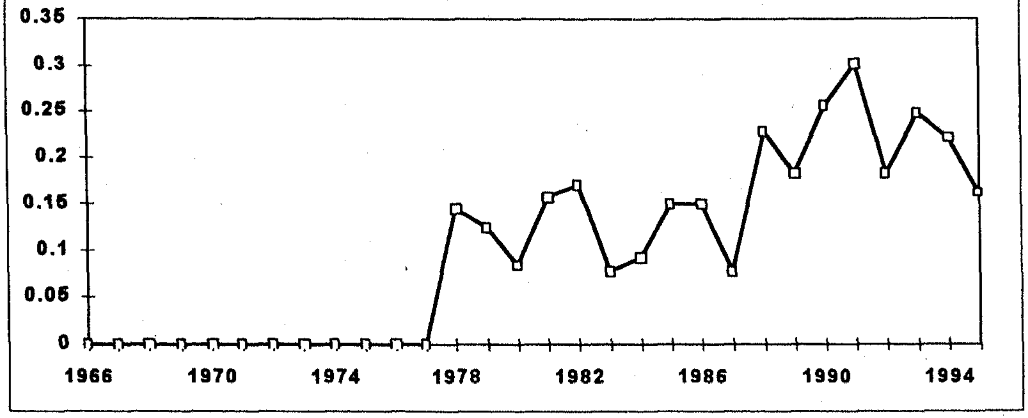


Figure A4.5 Model estimated proportion of neophyte nesting females, 1966-1995

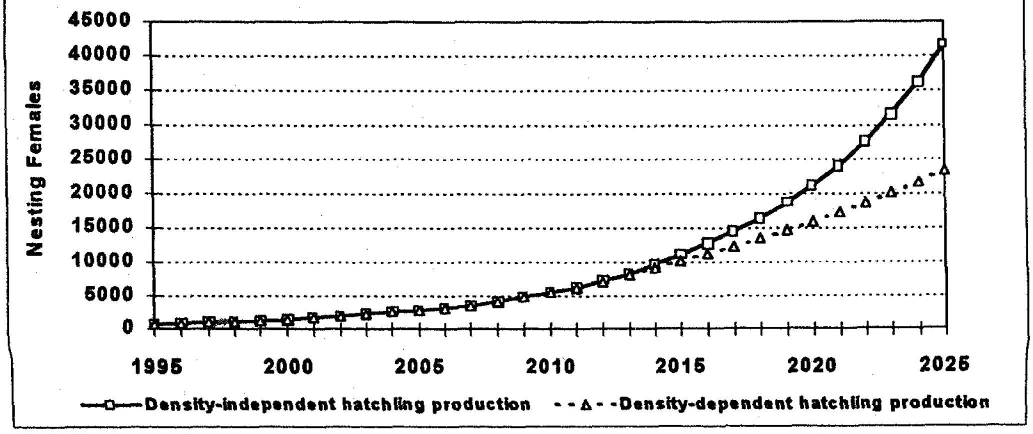


Figure A4.6 Model projections - estimated number of nesting females with and without density-dependent hatchling production

Appendix 4

Estimated
Female Standing Stock in 1995

Age	Number
0	63000.00
1	38429.86
2	21448.88
3	17314.81
4	10575.37
5	7609.96
6	5042.80
7	3468.69
8	1634.98
9	1174.96
10	807.03
11	603.93
12	224.75
13+	1173.32

ASSESSMENT OF THE LOGGERHEAD SEA TURTLE POPULATION (*Caretta caretta*) IN THE WESTERN NORTH ATLANTIC

A Report of the Turtle Expert Working Group
July 1, 1996
revised April, 1997

STOCK DEFINITION AND GEOGRAPHIC RANGE

The loggerhead sea turtle (*Caretta carreta*) occurs throughout the temperate and tropical regions of the Atlantic, Pacific, and Indian Oceans (Dodd, 1988). Its range of habitat includes open ocean waters, continental shelves, bays, lagoons, and estuaries. Loggerheads in the Western North Atlantic nest on high-energy, sandy beaches between the latitudes of 18° and 35° North. Worldwide, nesting on warm temperate beaches is much more common than nesting in the tropics.

Loggerhead turtles are best known at their nesting beaches. Nesting females are iteroparous and show both philopatry between nesting seasons and site fixity between successive nests within a season (Carr, 1967; Dodd, 1988). Also, there is genetic evidence that loggerhead females return to nest on the same region of coast where they first entered the water as hatchlings (Bowen *et al.*, 1993).

On average, loggerhead adult females in the southeast U.S. make reproductive migrations between foraging areas and nesting beaches every 2.5 yrs (Richardson and Richardson, 1982). On each migration, females deposit clutches in an average of 4.1 nests (Murphy and Hopkins, 1984) with each nest averaging approximately 115 eggs (Dodd, 1988).

In the western North Atlantic there are at least 4 loggerhead subpopulations[1] separated at the

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[1] These separations are based on analysis of mitochondrial DNA which a turtle inherits from its mother. Based on mtDNA, Cheloniid sea turtle species worldwide have highly structured nesting assemblages (Bowen *et al.*, 1992; Bass, 1994; Bowen *et al.*, 1994; Broderick *et al.*, 1994). Maternally mediated gene flow based upon mtDNA analysis among identified populations is low (Bowen *et al.*, 1993; Bowen, 1995). Natal homing is a dominant force shaping this phylogeographic structure. The nesting assemblages probably represent distinct genetic entities, but other research, including analysis of nuclear DNA (inherited from both parents) is necessary to address stock definition. Analysis of nuclear DNA of green turtles in both the Pacific and in the Atlantic found moderate rates of male-mediated gene flow (Karl *et al.*, 1992), but still, populations within ocean basins were structured. Nevertheless, given the high site fidelity of nesting females to their natal beach and low gene flow between nesting assemblages, most Western North Atlantic loggerhead nesting assemblages are vulnerable to extirpation.

nesting beach (Bowen, B. personal communication to Sandy MacPherson, October 28, 1995; Bowen *et al.*, 1993; Bowen, 1995; Encalda *et al.*, in press). Further investigation may delineate additional management units. Described by their geographic nesting distribution (Figure 1), these subpopulations are:

1. The Northern Nesting Subpopulation, occurring from North Carolina to northeast Florida, about 29° N. This subpopulation averaged about 6,200 nests/year, 1989-1995, on surveyed beaches (N.C. Dep. Environ., Health, and Nat. Resourc., unpubl. data; S.C. Dep. Nat. Resourc., unpubl. data; Ga. Dep. Nat. Resourc., unpubl. data; Fla. Dep. Environ. Protect., unpubl. data).

2. The South Florida Nesting Subpopulation, occurring from 29° N on the east coast to Sarasota on the west coast. This subpopulation averaged about 64,000 nests/year, 1989-1995, on surveyed beaches (Fla. Dep. Environ. Protect., unpubl. data). This is the largest loggerhead nesting assemblage in the Atlantic and is the second largest in the world (Ross, 1982).

3. The Florida Panhandle Nesting Subpopulation, occurring at Eglin Air Force Base and the beaches near Panama City. This subpopulation averaged about 450 nests/year, 1989-1995 (Fla. Dep. Environ. Protect., unpubl. data).

4. The Yucatán Nesting Subpopulation, occurring on the eastern Yucatán Peninsula, Mexico (Márquez, 1990). This subpopulation has about 1,500-2,300 nests/year (Zurita *et al.*, 1993; R. Márquez, unpubl. data).

Historically, nesting has been minor elsewhere in the western North Atlantic, except in Honduras, Nicaragua, Columbia, Venezuela, Bahamas, Cuba, Jamaica, and Puerto Rico (Sternberg, 1981). The present size of these nesting assemblages is unknown and individuals from these nesting areas have not been assayed for genetic affinity. As a whole, nesting by the loggerhead subpopulations of the northwest Atlantic account for 35-40% of the global nesting activity of this species (Ross, 1982).

Should an assemblage be extirpated, regional dispersal will not be sufficient to replenish the depleted nesting assemblage within thousands of years. This has been demonstrated amply through the loss of assemblages worldwide; they have not been re-established (*e.g.* Bermuda, Cayman Island, Alto Velo). Thus, both genetic evidence (mtDNA) and the historical record indicate that recolonization of extirpated nesting assemblages does not occur on contemporary time scales (Bowen *et al.*, 1994). For conservation and management, mtDNA data indicate that nesting assemblages must be considered independent demographically. This conclusion holds even if males should prove not to be philopatric to their natal site, because the production of progeny depends on female nesting success. For this reason, we use the term "subpopulation" to describe the nesting assemblages, consistent with the IUCN definition of the term.

Hatchlings from loggerhead nests show a brief, 1-3 day period of frenzied activity (crawling then swimming) that gets them to the water and away from land. Following this frenzy, loggerheads float inactively at the surface and become associated with minor and major oceanic convergence zones near current boundaries (Carr, 1987) and are carried by ocean currents across ocean basins. Loggerheads originating from U.S. Atlantic beaches are believed to lead a pelagic existence in North Atlantic Gyre for an extended period of time, perhaps as long as 10-12 years, and are best known from the eastern Atlantic near the Azores and Madeira (Maigret, 1983; Carr, 1986, 1987; Bjorndal *et al.*, 1994; Bolten *et al.*, 1994). We will refer to this life history stage as pelagic immatures. The distribution of pelagic immatures originating from Gulf of Mexico beaches is unknown. It is possible that they are carried by the Loop Current and Florida Current into the North Atlantic Gyre or are kept within the Gulf by eddies (Collard and Ogren, 1990).

At approximately 40-60 cm straight carapace length (SCL), loggerheads which have inhabited the surface waters of the open ocean begin recruiting to coastal areas where they become benthic feeders (Carr, 1987). Developmental habitat for these loggerheads includes lagoons, estuaries, bays, river mouths, and coastal waters typically less than 100 m deep. In U.S. waters small benthic immature turtles (<70 cm SCL) are the predominant loggerhead size class found everywhere from Cape Cod southward through Texas, except in southern Florida and the eastern Gulf of Mexico where large benthic immature loggerheads (70-91 cm SCL) are the most common size class (Medonça and Ehrhart, 1982; Lutcavage and Musick, 1985; Burke *et al.*, 1993; Teas, 1993; Quantum Resources, 1994; Epperly *et al.*, 1995b,c; Schroeder *et al.*, 1995) (Table 1, Fig. 2). Benthic immature turtles occupy coastal feeding grounds for a decade or more before maturing and making their first reproductive migration (Carr, 1987).

Genetic evidence has shown that assemblages of benthic immature loggerheads on foraging grounds comprise a mix of subpopulations. Loggerheads from Chesapeake Bay southward to Georgia are nearly equally divided in origin between the South Florida subpopulation and the Northern subpopulation. The representation of South Florida and Northern subpopulations is: Georgia, 41% and 59% (Sears, 1994); South Carolina, 50% and 50% (Sears *et al.*, 1995); Chesapeake Bay, 46% and 54% (Norrgard, 1995). In the Mediterranean, 57% of the immature loggerheads were from the South Florida subpopulation and 43% from the Mediterranean subpopulation (Laurent *et al.*, 1993; Bowen, 1995). The genetic origins of benthic immatures in the Gulf of Mexico have not been determined.

Adult loggerheads are known to make considerable migrations between foraging areas and nesting beaches (Dodd, 1988). Based upon the returns of tags applied at nesting beaches, non-nesting adult females from the South Florida subpopulation are distributed throughout the Bahamas, Greater Antilles, Yucatán, eastern Gulf of Mexico, and southern Florida (Bell and Richardson, 1978; Meylan, 1982; Meylan *et al.*, 1983). Likewise, non-nesting adult females from the Northern subpopulation occur along the east coast of the U.S. and occasionally in the northheastern Gulf of Mexico (Meylan, 1982). Limited tagging data suggest those adult females nesting in the Gulf of Mexico which are not part of the South Florida subpopulation remain in the Gulf of Mexico (Meylan, 1982).

There is very little information regarding the distribution of adult male loggerheads. Most strandings of adult males occur early in the nesting season, suggesting that following mating males move away from the nesting beaches. Two exceptions are known. In South Florida, adult males are observed year-round (NMFS, unpubl. data) and males involved in mating behavior are observed throughout the nesting season in Cape Lookout Bight, N.C. (Cape Lookout National Seashore, unpubl. data).

Both benthic immature and adult loggerheads in U.S. waters make seasonal migrations. Among loggerheads inhabiting the Atlantic coast, there is a spring migration of benthic immatures from south of Cape Hatteras northward and an autumn migration from the Middle Atlantic Bight southward (Shoop and Kenney, 1992; Epperly *et al.*, 1995b,c). In this autumn migration some animals move south to Florida waters (Keinath, 1993; Epperly *et al.*, 1995b,c) and others remain along the western wall of the Gulf Stream as far north as 35°47' north latitude (*Ibid.*). North of central Florida, there is also seasonal movement from offshore waters to nearshore and inshore waters in the spring and offshore movement during the fall and early winter (Epperly *et al.*, 1995b). Little is known about seasonal movement of loggerheads in the Gulf of Mexico.

POPULATION SIZE

Nesting data collected on index nesting beaches 1989-1995 represent the best dataset available to index the population size of loggerhead sea turtles. These data provide annual estimates of the number of nests laid each year and indirectly provide estimates both of the number of females nesting in a year (based on the average number of nests per nesting female, 4.1) and of the number of adult females in the entire population (based on the average remigration interval, 2.5). The total number of nests laid along the U.S. Atlantic and Gulf coasts, 1989-1995, has ranged from 53,016-85,306 annually, representing, on average, an adult female population of 43,060 (Table 2) [(nests/4.1) * 2.5]. Generally, about 90.5% of the nests were from the South Florida subpopulation, 8.8% were from the Northern subpopulation, and 0.6% were from the Florida Panhandle subpopulation.

Two methods were employed to estimate the current population size of all post-pelagic loggerheads in the coastal waters of the U.S. Atlantic and Gulf of Mexico coasts. These methods are described in detail in Appendices 1-2, along with assumptions and limitations of each method. Both employed ratio estimates using numbers of nests, strandings, and, in the second method, aerial survey data. A third method, a deterministic model, was deemed an unacceptable approach at this time due to the number of untenable assumptions and the lack of information on necessary input parameters (Appendix 4). The results of the two ratio methods are similar (Table 3); the mean post-pelagic loggerhead population size estimated for the 1989-1995 period was 224,321 and 234,355 turtles. These assuredly are underestimates for a number of reasons (see assumptions and implications in Appendices 1 and 2), especially because the methods only estimate the population in nearshore waters from which turtles are likely to strand. Based on aerial survey results (see Appendix 3), about 54% of the post-pelagic loggerhead population is in the southeastern U.S. Atlantic, 29% in the northeastern U.S. Atlantic, 12% in the eastern Gulf of Mexico, and 5% in the western Gulf of Mexico.

STATUS

The loggerhead was listed on July 28, 1978 as a threatened species under the Endangered Species Act of 1973 (PL93-205). The species is considered to be "Endangered" by the IUCN (the World Conservation Union) and is listed in Appendix I of CITES (Convention on International Trade in Endangered Species of Flora and Fauna).

In the Western North Atlantic, most loggerhead tagging, aerial surveys, and ground surveys of nesting began in the 1970's, prior to which there are no comparable data. However, commercial landings data indicate that loggerheads were probably much more abundant historically than they are today (Witzell, 1994). In the first recovery plan for the loggerhead turtle (Hopkins and Richardson, 1984), loggerhead abundance in the U.S. Atlantic and Gulf was estimated to be approximately 14,150 nesting females in 1983 (assuming an average of 4.1 nests per nesting female annually). If one assumes an average remigration interval of 2.5 years, an estimate of the total number of mature females in 1983 is calculated to be 35,375. A more recent (1989-1995) estimate of approximately 43,061 females is higher, in part, due to increased coverage of nesting beaches.

Despite some limitations, yearly numbers of nests remain the most widely available and reliable indicators of stability in loggerhead subpopulations. Principal limitations are that abundance is indexed to reproductive output and that only one sex of the oldest life-history stage in the population is represented. Consequently, nesting numbers respond sluggishly to changes in the benthic immature population and may be sensitive to changes in reproductive periodicity.

A principal advantage of using nesting data to track trends in abundance is that the data have become widely available in recent years. The most reliable nesting data are from complete censuses of nesting or from large unbiased survey-samples gathered in a standardized fashion. Surveys like this came to cover a large majority of southeast U.S. nesting beaches by 1989. Although time series are short for beaches with the most complete nesting-survey coverage, there are some individual beach sites within subpopulation ranges that have been surveyed extensively for longer periods of time.

Trends in the Northern Subpopulation

Two sites that provide the longest time series for assessing nesting trends in the Northern Subpopulation are: Cape Island, S.C. (surveyed completely since 1973; U.S. Fish Wildl. Serv., 1995; L. Klinek, pers. comm.) and Little Cumberland Island, Ga. (surveyed completely since 1964; J. Richardson, unpubl. data). During the period 1989-1995, annual nesting at Cape Island was found to predict the total nests for the Northern Subpopulation well (r^2=0.90, b=0.95, SE of b=0.14, P=0.001, n=7). However, a significant relationship could not be found between Little Cumberland Island nesting and nesting for the total subpopulation during the same period (r^2=0.30, b=0.55, SE of b=0.37, P=0.20, n=7).

At Cape Island, annual numbers of nests from 1973-1995 ranged from 579-2,790 with a significant decline averaging 3.2% per year (r^2=0.47, b=-0.69, SE of b=0.16, P=0.0003, n=23). Most of this decline appears to have occurred prior to 1979 after which a trend in nesting cannot be detected (r^2=0.00, b=-0.01, SE of b=0.26, P=0.97, n=17). Hopkins-Murphy and Murphy (1988) examined statewide nesting in South Carolina and found an average 5% per year decline in nesting between the period spanning 1980-1982 and the period of 1985-1987 (three-year periods that are believed to encompass the majority of nesting in the subpopulation). Strandings declined at a similar rate during this period (Crowder *et al.*, 1995), which may indicate a decline in the total number of benthic loggerheads.

Although recent nesting at Little Cumberland Island did not mirror nesting of the Northern Subpopulation as a whole, trends in the long time-series of nesting data there may reflect occurrences that shorter time series do not. During the period 1964-1995, nesting at Little Cumberland Island ranged from 34-278 with a significant decline averaging 2.6% per year (r^2=0.67, b=-0.82, SE of b=0.10, P<0.0001, n=32). A significant decline, 2.5% per year, also is seen for the period 1964-1988 (r^2=0.47, b=-0.69, SE of b=0.15, P=0.0002, n=25) but cannot be demonstrated for the period 1989-1995 (r^2=0.21, b=-0.46, SE of b=0.40, P=0.30, n=7).

Extensive standardized aerial, ground, and tagging surveys conducted throughout the Northern subpopulation's nesting range have begun only recently around 1989 (Meylan et al., 1995; N.C. Dep. Environ., Health, Nat. Resourc, unpubl. data; S.C. Dep. Nat. Resourc., unpubl. data; Ga. Dep. Nat. Resourc., unpubl. data; Fla. Dep. Environ. Protect., unpubl. data). During the period 1989-1995, annual numbers of nests ranged 4,372-7,899 with no detectable trend (r^2=0.017, b=0.13, SE of b=0.44, P=0.78, n=7).

Trends in the South Florida Subpopulation

In the South Florida Subpopulation range, there are eight beaches that have been surveyed completely since 1983 (Meylan et al., 1995; Fla. Dep. Environ. Protect., unpubl. data). One of these beaches, Hutchinson Island, was surveyed biannually, 1971-1981, and annually, 1981-1994 (Quantum Resources, 1994). Annual nesting from a composite of the eight Florida beaches predicted annual nesting at all Florida index beaches well (r^2=0.96, b=0.98, SE of b=0.10, P<0.001, n=7) as did annual nesting only at Hutchinson Island (r^2=0.96, b=0.98, SE of b=0.10, P<0.001, n=7). This indicates that for the period prior to complete coverage of the nesting range, both Hutchinson Island and the composite of beaches may accurately reflect nesting trends for the total South Florida Subpopulation.

At Hutchinson Island during the period 1971-1994, nesting ranged from 2,757-7,126 with a significant trend of 4.0% per year (r^2=0.62, b=0.79, SE of b=0.15, P<0.0001, n=19). Trends for Hutchinson Island were also significant for the period 1971-1988 (2.2% per year, r^2=0.32, b=0.56, SE of b=0.25, P=0.04, n=13) and for the period 1983-1994 (4.5% per year, r^2=0.64, b=0.80, SE of b=0.19, P=0.002, n=12). The composite of eight beaches also showed a significant increase in nesting (5.7% per year; range, 14,627-26,776 nests per year) for the 1983-1994 period (r^2=0.63, b=0. 80, SE of b=0.19, P=0.002, n=12).

As in the Northern subpopulation range, extensive standardized nesting surveys in Florida have begun only recently. The most complete and unbiased indication of annual nesting numbers are for the period 1989-1995 and come from censuses on a set of index beaches that account for approximately 80% of the State's nesting (Schroeder, 1994; Fla. Dep. Environ. Protect., unpubl. data). The number of nests on index beaches for this period ranged 39,092-59,147 with no detectable trend (r^2=0.33, b=0.58, SE of b=0.37, P=0.18, n=7).

One of very few unbiased indices of abundance for benthic immature and adult loggerheads can be found on the south-central Atlantic coast of Florida: the Florida Power and Light Company, St. Lucie Nuclear Power Plant. There, turtles are drawn into a cooling water intake that has operated almost continously since 1976 (Quantum Resources, 1994). Captures at the plant from 1977-1995 reveal cycles in abundance for all size classes (Fig. 3) (Quantum Resources, unpubl. data). During this period, numbers of adult loggerheads ≥ 92 cm SCL increased significantly (r^2=0.52, b=0.72, SE of b=0.17, P=0.0005, n=19), but a trend cannot be demonstrated in numbers of small benthic immatures < 70 cm SCL (r^2=0.10, b=0.31, SE of b=0.23, P=0.20, n=19) nor in numbers of large benthic immatures 70-91 cm SCL (r^2=0.18, b=0.43, SE of b=0.22, P=0.07, n=19). As in the NMFS stranding data for this region few adult loggerheads occur outside the nesting season.

Trends in the Florida Panhandle and Yucatán Subpopulations

Although nesting surveys in the Florida panhandle and in the Yucatán Peninsula have been too irregular to allow a meaningful analysis of trends, recent surveys do provide estimates of current nesting levels. Annual nesting in the Florida panhandle 1989 to 1995 ranged from 113-928 nests (Meylan *et al.*, 1995; Fla. Dep. Environ. Protect., unpubl. data) and annual nesting in the states of Campeche, Yucatán, and Quintana Roo, Mexico is estimated to be roughly 50, 100, and 1,331-2,166 nests, respectively (Zurita *et al.*, 1993; Rene Marquez, unpubl. data). Nesting in Quintana Roo appears to be declining (R. Márquez, unpubl. data). Elsewhere, Juventud Island, Cuba, is believed to have nesting on the order of several hundred nests per year (Rene Marquez, unpubl. data).

Status of Subpopulations

Although there are indications that numbers of loggerheads in the western North Atlantic have been reduced, recent trends can be assessed only for adults of the Northern and South Florida Subpopulations. Best-available nesting data reveal that these two subpopulations have followed different trends in abundance. The Northern Subpopulation has declined substantially in the last 23 years but may have stabilized in recent years. At least one nesting assemblage within the Northern subpopulation range (Little Cumberland Island) has shown alarming declines in the last 32 years, although it is not clear how these declines parallel nesting of the Northern subpopulation as a whole.

Adult loggerheads of the South Florida Subpopulation have shown significant increases over the last 25 years indicating that the population is recovering. However, we were unable to detect a significant trend over the most recent 7 years of nesting - a short time span for these types of analyses. An increase in numbers of adult loggerheads in the water off eastern Florida in the last 20 years appears to match trends seen in nesting numbers for the same extended period. However, numbers of immatures in the same area have not increased. If these data are indicative of loggerhead abundance elsewhere, limited recruitment to adulthood can be expected to foreshorten the recovery of the South Florida Subpopulation measured at the nesting beach. Because loggerheads take approximately 20-30 yr to mature, reduced numbers of benthic immatures would not be expected to result in nesting beach declines for 10-20 yr. The effects of pelagic-stage mortality (*e.g.*, from Atlantic and Mediterranean longline fisheries) would take even longer to become evident on nesting beaches. Status assessments from nesting data should be made cautiously with this in mind.

ANNUAL HUMAN-CAUSED MORTALITY AND SERIOUS INJURY

Eggs, hatchlings and small juvenile loggerheads are subject to nest disturbances and predation, resulting in high natural mortality rates, which are believed to decline with increasing body size. In recent decades, however, an additional suite of human-induced, size-independent, mortality factors have taken their toll. National Research Council (1990) summarized these (Table 4), and concluded that commercial fishing, in particular bottom trawls in the southeastern shrimp fishery, was by far the most important mortality source, killing more immature and adult turtles than all other human sources combined. Quantification of these mortality sources, and particularly their impacts on population dynamics, have been much harder. Henwood and Stuntz (1987) estimated an annual lethal take of 9,874 loggerheads in the U.S. Atlantic and Gulf of Mexico shrimp fisheries. National Research Council (1990) resorted to estimating an order of magnitude range, concluding that shrimp trawling drowned 5,000 - 50,000 loggerheads annually in southeastern U.S. waters, prior to the use of turtle excluder devices. They suggested that the estimates of Henwood and Stuntz could be underestimates by a factor of 3-4.

Stage-based matrix population models developed for loggerheads in the Southeast (Crouse *et al.*, 1987; Crowder *et al.*, 1994) have used sensitivity analyses to evaluate the relative contribution of various life history stages and parameters (age to maturity, fecundity, stage-specific mortality rate, *etc.*) to overall population growth. They found that the model population growth rate was relatively insensitive to changes in fecundity and survival of the egg/hatchling stage. However, they found that the model population was quite sensitive to changes in annual survival of the larger (older) stages, particularly 50 - 80 cm SCL benthic immatures. This is also the predominant size class represented in U.S. loggerhead strandings (Crouse *et al.*, 1987; Teas, 1993). Thus, Crouse *et al.* (1987) concluded that reduction in human-induced mortality of benthic immature and adult loggerheads was likely the most effective ways to effect recovery of depleted populations.

Based on these and other findings, NMFS began requiring TEDs in some shrimp trawls in late 1989. By January 1993 TEDs were required in most southeastern shrimp trawls with the remainder coming on line in December 1994 (57 FR 57348, December 4, 1992). The use of TEDs has reduced trawling-related mortality of loggerheads. Crowder *et al.* (1995) reported a 44% overall reduction in loggerhead strandings in South Carolina coincident with the implementation of TED requirements. Recently, the National Marine Fisheries Service estimated that, based on observer data (36 takes observed with 0.2% observer coverage over 4 years), the total take of the U.S. shrimp fishery has been approximately 4,500 loggerhead sea turtles annually (11 June 1996 Biological Opinion). If one assumes all these takes were lethal and compares that estimate to Henwood and Stuntz's estimate of mortality for the 1980's, TEDs have reduced lethal takes by about 54% (85-95% if Henwood and Stuntz's estimates were low by a factor of 3-4). Also, sea turtle nesting in South Carolina, which had been experiencing a decline of 5% annually through the mid-1980's (Hopkins-Murphy and Murphy, 1988), appears to have stabilized subsequent to the implementation of TEDs (Hopkins-Murphy and Murphy, 1994; Hopkins-Murphy, unpubl. data).

Other mortality sources that are being addressed also include:

- The summer flounder bottom-trawl fishery off southern Virginia and North Carolina. Observer data indicated catch-per-unit effort rates (CPUE) equal to or exceeding those of the shrimp fishery (Epperly *et al.*, 1995c), although the fishing effort was much lower, resulting in fewer overall takes. TEDs have been required in this fishery since 1992. Additionally, observers are currently placed on summer flounder trawlers as far north as New York to ascertain whether TEDs should be required elsewhere in this fishery.

- Dredging. The use of hopper dredges is seasonally restricted in southeast U.S. Atlantic waters to minimize potential impacts on sea turtles. All hopper dredges used in this area must have full-time observer coverage.

- Oil-platform removals. The Minerals Management Service requires a precise set of procedures to ensure that sea turtles are not impacted by detonation of charges used in platform removal.

- Beach nourishment. Seasonal restrictions, sand compatibility and even sand tilling are required for various beach nourishment projects throughout the southeast.

- Nest protection: a whole host of measures, including restricting vehicular access, nest screening (against predators), nest relocation (erosion), regulating beachfront lighting (nesting female and hatchling disorientation) and others are pursued to various degrees at various beaches to increase nest success.

- Longline fisheries. The Crouse *et al.* (1987) loggerhead population model was sensitive to changes in mortality of the pelagic immature stage loggerheads. It is known that pelagic longline fisheries do take turtles, sometimes in large numbers (Aquilar *et al.*, 1992; Brady, 1994; Witzell, 1984). While protection of this stage is much more difficult, NMFS has begun consultation regarding U.S.-based longline fisheries in both Atlantic and Pacific waters.

- Gill net fisheries. There are a number of more or less "local" gill net fisheries (*e.g.*, pompano in east central Florida, black drum at the mouth of the Chesapeake Bay, inshore flounder in North Carolina, *etc.*), ranging the length of the U.S. Atlantic and Gulf coasts which take turtles to a larger or smaller degree. NMFS has been working with the states involved in an attempt to identify and address these gill net takes, but progress has been slow to date.

- Power plant intakes. Several power plants have been found to entrain sea turtles through their cooling water intake tubes. Screens have been helpful at some; others monitor regularly to release entrained turtles. Most turtles are tagged and released unharmed. These locations have been useful to obtain live turtles for a number of studies.

- Recently NMFS has recommended that the U.S. Coast Guard consult with NMFS regarding the potential impacts of generic permits for speed boat races, regattas, *etc.*, and their potential impacts on endangered and threatened sea turtles.

INTERIM STRANDING LIMITS

While undertaking to reduce mortality from all sources, NMFS has been unable thus far to systematically assess cumulative human-induced mortalities and their effects on sea turtle species' continued existence and recovery. Understanding the population response to changes in various life history parameters is prerequisite to establishing Interim Strandings Limits for human-induced mortality factors. Consequently, this was one of the primary tasks the Turtle Expert Working Group was asked to undertake.

Currently, the primary index of turtle mortality comes from sea turtle strandings (turtles that wash up on monitored beaches and are reported to the Sea Turtle Stranding and Salvage Network) (Table 5). The vast majority (90%) of these strandings are dead animals. Two shortcomings of this measure of mortality are that: (1) it is most sensitive to nearshore mortality and completely insensitive to mortalities far offshore; and (2) the percentage of carcasses that wash ashore probably is dependent on wind and ocean currents, *etc.* The majority of strandings show no obvious cause of death, such as injuries from a boat propeller or shark attack, entanglement in fishing net, emaciation suggesting illness, *etc.* What can be said, is that most stranded turtles appeared healthy at the time of death, many were actively feeding and have full stomachs; their good health and lack of obvious injuries is consistent with drowning in shrimp trawls although other factors may contribute to some of these deaths, also.

It is difficult to determine what proportion of at-sea mortality is reflected in strandings. Murphy and Hopkins-Murphy (1989) released marked carcasses offshore South Carolina, of which only 28% were later recorded as strandings. Epperly *et al.* (1996) reported that, at most, 7-13% of the mortality attributable to the winter trawl fishery for summer flounder stranded. Another rough estimation can be made by noting that the National Academy of Sciences panel estimated that 5,500 - 55,000 (most likely around 33,000-44,000) loggerheads and Kemp's ridleys drowned annually in shrimp trawls in the late 1980s prior to TED requirements (National Research Council, 1990). In the years 1986-1989 (when the stranding network monitoring effort was both high and consistent), total strandings for all species and all mortality factors reported by the stranding network ranged from 1,991 - 2,373. This suggests that approximately 5-6% (range 3.6 - 43.1%) of the total mortality due to shrimp trawls was reflected in strandings over this period. In other words, while strandings in the presence of shrimp trawling can be used as a rough index of trawling mortality, they likely underestimate trawling mortality, severely at times.

A number of studies have shown that, at least at certain places and times (primarily Texas and South Carolina), strandings, in number, have been significantly associated with offshore shrimp trawling effort (Caillouet *et al.*, 1991; Caillouet *et al.*, 1996; Crowder *et al.*, 1995; Gallaway *et al.*, 1995, National Research Council, 1990). The National Research Council (1990) concluded that, at certain times of the year in South Carolina and Texas, 70 - 80% of strandings were due to shrimp fishing mortality. Crowder *et al.* (1995) assessed a 14 year time period of one of the most consistent and reliable strandings datasets available (South Carolina) and found a statistically significant correlation between shrimp trawling activity and strandings.

The Turtle Expert Working Group (TEWG) concludes that estimation of the maximum number of individual loggerheads that can be taken incidentally to commercial fishing cannot be made at this time. The estimation of current total population size, mortality rates and population growth rates, would be unreliable at this time, due to deficiencies in data and model sructure. Furthermore, because of the long pelagic stage and delayed age of maturity of loggerheads, using nesting trends to determine population status is much riskier than with a faster maturing species, such as Kemp's ridleys.

Despite the shortcoming of using strandings as an index of turtle mortality, the TEWG judged that strandings remain the best data currently available for estimating nearshore turtle mortality. For the next year, the TEWG recommends use of a proxy Interim Strandings Limit (ISL) based on the previous five-yr running average of strandings, plus one standard deviation to account for annual variation (Table 6). While the TEWG recommended indexing a similar calculation by the rate of increase of nesting for Kemp's ridley's, the group does not recommend such an adjustment for loggerheads for several reasons:

(1) The rate of increase of nests for the South Florida subpopulation, about 4%, is much lower than that for Kemp's. The rate is not statistically different from zero for the period 1989-1994.

(2) There has been no detectable increase in nesting in the Northern subpopulation.

(3) Due to the slow maturation rate in loggerheads, the period over which pelagic and benthic immatures are exposed to mortality may be as long as 15 or more years, making it risky to use annual numbers of nests as indices of annual population size or changes in population size. For example, it is known that pelagic juvenile loggerheads are being taken, potentially in large numbers, in several high seas longline fisheries (Aquilar *et al.*, 1992; Brady, 1994; Witzell, 1984). It is believed also that fishing effort in these fisheries has increased considerably since the United Nations high seas driftnet ban went into effect in December 1992. The impacts of such losses will not be reflected in nesting numbers for as many as 20yr or more. This latter reason also makes it imperative that a better method of deriving and indexing an ISL be developed over the next year.

Once an ISL has been calculated, the takes must then be allocated among potential "user" groups. This task is not in the charge of the TEWG, but the group has some recommendations with regard to this:

(1) When an obvious natural mortality cause, such as cold-stunning, is identified, the Group recommends removing those individuals from the ISL tally. The ISL is derived in such a way as to identify potential strandings above natural mortalities, and if these were included, they would inappropriately increase the 5-year running average used for calculating future years' ISLs;

(2) For management purposes, it likely would be beneficial to calculate and monitor the ISLs on a regional and seasonal basis, allowing better assessment of whether current mortality is unusual and excessive.

It can be seen (Table 6) that the ISLs for loggerheads calculated in this manner were exceeded in the western Gulf of Mexico in 1994 and in the northeast U.S. Atlantic in 1992, 1993, and 1995, and nearly equalled it in 1994 in this latter area. These excesses in mortality relative to historical levels, along with the information that the combined subpopulation have shown no signs of increase overall, strongly suggests the need to reduce mortality for this species.

Finally, the TEWG underscores the temporary nature of this interim proxy (ISL): as noted above, strandings are clearly an underassessment of actual nearshore mortality and do not measure offshore mortality at all; the long time to maturity for loggerheads makes basing any take limits on nesting numbers very risky. Furthermore, this interim proxy (ISL) may be to high, which could be detrimental if used beyond 1996 (*i.e.*, in the northeast, where ISLs have been exceeded in three of the last four years, those years of excessive strandings now form the basis for calculating the mean, and therefore, next year's ISL). Both additional information on juvenile population abundance and population trends, and additional work on deriving a more suitable, less risky, ISL calculation are necessary.

The TEWG did examine the approach codified under the 1994 reauthorization of the Marine Mammal Protection Act, where takes by fisheries and other human activities are now managed by calculating a Potential Biological Removal (PBR) (Barlow *et al.*, 1995). PBR is that number of animals that can be removed from the population by human activities while still allowing the population to recover, and move toward Optimum Sustainable population size. PBR is calculated from the minimum population size and the intrinsic rate of growth of the population, with conservative factors built into the calculation in several ways thus reducing the potential for overestimating allowable removals. The PBR calculation was validated by a series of simulations of possible marine mammal population trajectories, to determine if they would be adequately protected.

Calculating a minimum population size for the western North Atlantic loggerhead assemblages required a number of assumptions, many of which were violated. Furthermore, the particular equation for calculating PBRs was validated for populations with a marine mammal-like life history which might be inappropriate for marine turtles. The TEWG plans to pursue development of a similar approach to management of human mortality factors for endangered and threatened sea turtles over the next year by attempting to derive and validate an ISL calculation based on similar principles for sea turtles.

CONCLUSIONS

1. There is evidence supporting the separation of Western North Atlantic loggerhead turtles into at least four distinct subpopulations, three within the continental U.S. They cannot yet be managed separately because we do not know the in-water distribution of each subpopulation.

Research suggests that at least two of the subpopulations intermingle on the foraging grounds of the U.S. Atlantic coast.

2. Given the recovery goal of more than 12,800 nests for the Northern Subpopulation (inferred from the recovery plan) and given that currently nests number about 6,200 and have not increased perceptibly, and that the number of nests declined during the 1980's, it appears unlikely that the recovery goal for this subpopulation will be achieved.

3. Given a recovery goal of measurable increases in the South Florida Subpopulation, and given an observed increase throughout the 1980's at eight Florida nesting beaches, it appears that this subpopulation is stable and may be increasing. However, the time frame for nesting surveys on most beaches utilized by this subpopoulation is too small to evaluate trends throughout the region. Thus, recovery rates for the entire subpopulation cannot be determined with certainty at this time. Further caution is warranted with this subpopulation because, although nesting activity has been increasing, catches of benthic immature turtles at the St. Lucie Nuclear Power Plant intake canal have not been increasing.

4. Given that a recovery goal was not set for the Florida Panhandle Subpopulation and that it is extremely small, beaches within this region need to be established as index nesting beaches to assess trends in this subpopulation.

5. Given that recovery goals were not set for the Yucatán Subpopulation nor for any other nesting assemblage in the Caribbean, steps need to be taken to assess the status of these groups. The Yucatán subpopulation appears to be declining.

6. The TEWG concludes that estimation of the maximum number of individual loggerheads that can be taken incidentally to commercial fishing operations cannot be made at this time. The TEWG was not able to model the population due to data and model structure deficiencies. The Group recommends use of a 1-yr proxy Interim Strandings Limit based on the previous five-yr running average of strandings, plus one standard deviation. During the interim, research to address the recognized data and model structure deficiencies should be initiated and a turtle-specific PBR approach should be examined to be used until sufficient data are available to model the population.

7. NMFS needs to assess all human-induced mortality sources and reduce their impact. In particular NMFS should address problems in the northeast U.S. Atlantic, where ISLs have been exceeded 3 of the last 4 yr and were nearly equal during the fourth year.

8. There are a number of Priority 1 recovery plan tasks that are NMFS' responsibility that have yet to be fully implemented. Several would address research recommendations listed in the next section.

RECOMMENDATIONS

Many of the following recommendations will produce data necessary to improve the estimates of population and assessment models being generated and analyzed by the Turtle Expert Working Group. Reduction of the level of uncertainty in these parameters will significantly increase the value of models and the accuracy of model results. NMFS needs to facilitate the completion of these recommendations.

Research Recommendations

1. Long-term, in-water indices of loggerhead abundance in coastal waters are needed to identify relative abundance of sea turtles over time; and to detect changes in size composition with implications regarding recruitment. Ongoing studies in St. Lucie and Indian River, Fla.and Pamlico Sound, N.C. should be continued. Additional studies should be started in other parts of the range, including one in each Sea Turtle Conservation Zone . All turtles caught should be tagged with PIT tags as well as flipper tags, and should be scanned for existing PIT tags. CPUE studies should be standardized by year to provide long-term comparability. The projects should be consistent and long-term, with data collected in a statistically defensible manner; this implicitly requires NMFS oversight and funding. Sea turtles encountered during these studies should be sampled for genetic material to support mtDNA and nuclear DNA studies. Habitat use and tracking studies conducted as adjuncts to abundance index studies should not compromise the effectiveness of the primary goal of indexing abundance. Projects which provide an assessment of trends in juvenile abundance should be given priority, to provide information on trends that will not be reflected immediately on nesting beaches.

2. Information is needed regarding the location and source of mortality of stranded sea turtle carcasses, as well as predictions regarding whether sources can be monitored by documenting strandings. Trajectory modeling of oceanographic conditions (considering decomposition rates and scavenging) should be developed to determine sea turtle carcass landfall patterns or to identify conditions under which carcasses would not be likely to strand.

3. Age and growth studies need to be completed.

4. Physiological studies to determine the effects of forced and repeated submersion should be continued.

5. Genetic studies of benthic immature and adult loggerheads stranded and collected in the Western North Atlantic should be conducted to determine natal origin.

6. The utility of population models could be increased through improved age-class survival estimates, improved information on the distribution of each subpopulation, and improved information on basic life history, such as (a) the number and distribution of number of nests laid

by a nesting female during a season and the distribution of re-migration intervals, (b) age and growth data to determine length in pelagia and age of maturity, *etc.*, (c) in-water abundance trends, and (d) sex ratios. Data collection from nesting beach activities, stranding monitoring, and in-water abundance surveys should support this goal.

Management Recommendations

1. Stranding network index areas within the southeast region should be developed for long-term standardized coverage by the STSSN. The statistical utility and defensibility of the index areas should be primary considerations in the identification and monitoring of stranding index areas.

2. Standardized PIT tag techniques and equipment needs to be developed. A centralized sea turtle PIT tag database should be developed. PIT tag scanners should be issued to representative STSSN participants in each state to maximize the mark-recapture components of long-term sea turtle studies, including nesting beach surveys.

3. NMFS' efforts to obtain funding to expand the observer program should continue. NMFS should work with industry to assess sea turtle CPUE in various areas throughout the Western North Atlantic, including inshore embayments, nearshore waters and offshore waters, and in the Eastern North Atlantic.

4. Attempts to reduce effort should continue. Limited time closures, gear restrictions, or other measures could be used effectively to reduce nearshore shrimping effort for increased sea turtle protection. Based on recommendations from these analysis, proposed rules implementing these recommendations should be promulgated in the event current protection measures are insufficient as evidenced by stranding levels exceeding the proposed ISLs, and strandings are determined to be caused primarily by shrimping activities.

5. NMFS should continue to evaluate the impacts of other sources of mortality, including gillnets, crab trawling, and bottom trawl fisheries, especially those that have high nearshore effort, and reduce all human-induced sources of mortality.

6. A loggerhead recovery team should convene and consider new information, especially the genetic data that indicate there are at least 4 subpopulations in the western North Atlantic. Recovery goals for each need to be defined.

7. The TEWG or another similar group should meet annually to identify and review the status of this species to ensure recovery.

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Table 1. Size frequency of post-pelagic loggerhead sea turtles stranding along the U.S. Atlantic and Gulf of Mexico coasts, by zone and size class, 1986-1994. Note that an additional number of turtles stranded that either were not measured or measurements were not standardized.

Statistical Zone		Straight Carapace Length		
	40.0-69.9 cm	70.0-91.9 cm	≥92.0 cm	Total
1	8	50	13	71
2		1		1
3	1	7	1	9
4	17	165	108	290
5	14	207	124	345
6	4	11	5	20
7	2	8	4	14
8	20	48	17	85
9	6	11	6	23
10	9	17	10	36
11	20	49	15	84
12		5	3	8
13	2	2	1	5
14	5	2		7
15				0
16				0
17	20	15	1	36
18	139	54	7	200
19	97	57	6	160
20	220	124	13	357
21	90	65	4	159
24	6	28	7	41
25	22	35	8	65
26	36	75	57	168
27	161	229	88	478
28	234	235	133	602
29	402	202	79	683
30	977	245	82	1304
31	462	187	115	764
32	456	165	80	701
33	215	113	62	390
34	323	107	55	485
35	256	97	44	397
36	64	57	24	145
37	30	2	2	34
38	27	24	10	61
39	51	25	15	91
40	93	14	5	112
41	75	2		77
42	5			5
43				0
44				0

Table 2. Loggerhead sea turtle nests in the U.S., 1989-1995. Minor nesting occurring north of North Carolina and west of Florida is not included. Data are from Meylan *et al.* (1995), Schroeder (1994), Maley (1995), Maley and Murphy (1994, 1993), Maley and Harris (1992, 1991), Harris and Maley (1990), Florida Department of Environmental Protection (unpubl. data), Georgia Department of Natural Resources (unpubl. data), South Carolina Department of Natural Resources (unpubl. data) and North Carolina Department of Environment, Health and Natural Resources (unpubl. data).

Area	Year							Mean 1989-1995
	1989	1990	1991	1992	1993	1994	1995	
Northern Nesting Subpopulation:								
North Carolina	459	815	931	731	487	1021	662	729
South Carolina	2444	4491	3657	3943	2757	4136	2959	3484
Georgia	691	1085	1209	1054	475	1375	1028	988
N.E. Florida	778	1355	952	759	809	1367	1149	1024
Total	4372	7746	6749	6487	4528	7899	5798	6225
Nesting Females	1066	1889	1646	1582	1104	1927	1415	1519
South Florida Nesting Subpopulation:								
S.E. Florida	46295	62071	63416	59677	50616	64219	71765	59723
S.W. Florida	2236	3085	3959	4186	3836	5395	6815	4216
Total	48531	65156	67375	63863	54452	69614	78580	63939
Nesting Females	11837	15892	16433	15576	13281	16979	19166	15595
Florida Panhandle Nesting Subpopulation:								
Total	113	174	287	351	560	773	928	455
Nesting Females	28	42	70	86	137	189	226	111
Total U.S. Nests	53016	73076	74411	70701	59540	78286	85306	70619
Total U.S. Nesting Females	12931	17823	18149	17244	14522	19094	20807	17224

Table 3. Comparison of post-pelagic loggerhead population sizes in the Western North Atlantic, 1989-94, calculated with ratio methods. Consult Appendices 1-2 for details on the methods.

Year	Ratio Method All Zones	Ratio Method S.E. U.S.A. with Spatial Expansion
1989	269,396	316,113
1990	242,707	233,010
1991	199,639	207,428
1992	203,503	183,627
1993	200,657	255,014
1994	236,675	254,827
Mean 1989-94	224,321	234,355

Table 4. A qualitative ranking of the relative importance of various mortality factors on juveniles or adults, eggs, and hatchlings with an indication of mortality caused primarily by human activities, and order-of- magnitude estimates of human-caused mortality on juvenile to adult loggerhead sea turtles. Table modified from National Research Council (1990).

		Life Stage			
Source of Mortality	Primarily Human Caused	Juveniles to Adults	Eggs	Hatchlings	Mortality (number/year)
Shrimp trawling	yes	high	none	unimportant	5,000-50,000
Other fisheries	yes	medium to low	none	unimportant	500-5,000
Non-human predators	no	low	high	high	
Weather	no	low	medium	low	
Beach development	yes	low	medium	low	
Disease	no	low	unimportant	low	
Dredging	yes	low	unimportant	unimportant	50-500
Entanglement	yes	low	unimportant	low	500-5,000
Oil-platform removal	yes	low	none	unimportant	10-100
Collisions with boats	yes	low	none	unimportant	50-500
Directed take	yes	low	medium	unimportant	5-50
Power plant entrainment	yes	low	none	unimportant	5-50
Recreational fishing	yes	low	none	unimportant	
Beach vehicles	yes	low to unimportant	medium	unimportant	
Beach lighting	yes	low to unimportant	unimportant	medium	
Beach replenishment	yes	unimportant	low	low	
Toxins	yes	unknown	unknown	unknown	
Ingestion of plastics, debris	yes	unknown	none	unknown	

Table 5. Total loggerhead strandings reported to the Sea Turtle Stranding and Salvage Network (STSSN) by region, 1980-1995.

Year	Western Gulf of Mexico (Zones 13-21)	Eastern Gulf of Mexico (Zones 1-12, partial 24&25)	Southeast U.S. Atlantic(Partial zones 24&25-35)	Northeast U.S. Atlantic (Zones 36-44)	Total
1980	55	38	1645	219	1957
1981	61	31	840	78	1010
1982	114	55	1071	101	1341
1983	52	49	595	154	850
1984	107	77	509	115	808
1985	84	89	603	80	856
1986	168	125	822	101	1216
1987	164	216	1187	198	1765
1988	125	186	903	168	1382
1989	113	295	887	143	1438
1990	147	163	1075	212	1597
1991	76	131	610	169	986
1992	70	120	702	231	1123
1993	72	127	563	229	991
1994	203	105	798	227	1333
1995	130	132	866	416	1544

Table 6. Strandings and Incidental Strandings Limits (ISL), by year, by region. western Gulf of Mexico, eastern Gulf of Mexico, southeast U.S. Atlantic, and northeast U.S. Atlantic. The ISL is the 5 year mean + 1 standard deviation (SD). For example, the ISL for the western Gulf of Mexico in 1996 is 163 stranded loggerhead turtles >40 cm that are not cold stun kills. This is calculated as the 5 year mean, ending with 1995. Asterisks indicate years when ISL was exceeded.

Year	Total Strandings	5 Year Mean	Standard Deviation	ISL Mean + 1SD
Western Gulf of Mexico: Zones 13-21				
1986	147			
1987	157			
1988	118			
1989	109			
1990	137	133.60	19.92	153.52
1991	73	118.80	31.56	150.36
1992	66	100.60	30.24	130.84
1993	67	90.40	31.48	121.88
1994*	198	108.20	58.33	166.53
1995	128	106.40	57.35	163.75
Eastern Gulf of Mexico: Zones 1-12, part 24 & 25				
1986	121			
1987	216			
1988	185			
1989	289			
1990	157	193.60	63.81	257.41
1991	122	193.80	63.53	257.33
1992	112	173.00	71.02	244.02
1993	121	160.20	74.02	234.22
1994	105	123.40	20.03	143.43
1995	129	117.80	9.36	127.16
Southeast U.S. Atlantic: Zones 26-35, part Zones 24 & 25				
1986	809			
1987	1159			
1988	887			
1989	874			
1990	1065	958.80	146.75	1105.55
1991	599	916.80	214.67	1131.47
1992	689	822.80	182.61	1005.41
1993	534	752.20	216.65	968.85
1994	758	729.00	206.31	935.31
1995	834	682.80	120.12	802.92
Northeast U.S. Atlantic: Zones 36-42				
1986	96			
1987	172			
1988	166			
1989	134			
1990	206	154.80	41.63	196.43
1991	158	167.20	26.06	193.26
1992*	205	173.80	31.24	205.04
1993*	220	184.60	36.73	221.33
1994	220	201.80	25.54	227.34
1995*	296	219.80	49.61	269.41

Figure 1. Idendified loggerhead nesting assemblages in western North Atlantic.

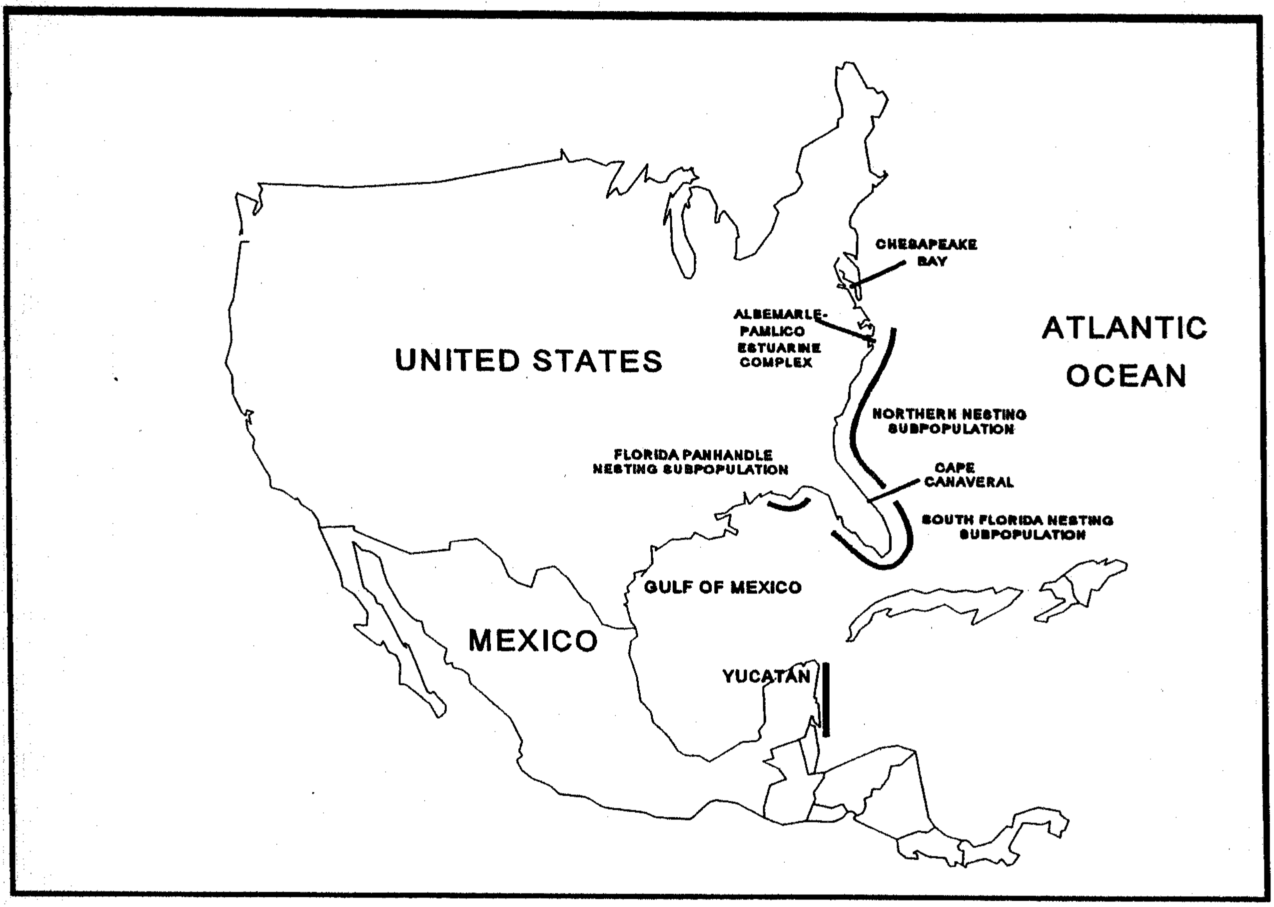


Figure 2. NMFS statistical zones used to report sea turtle strandngs.

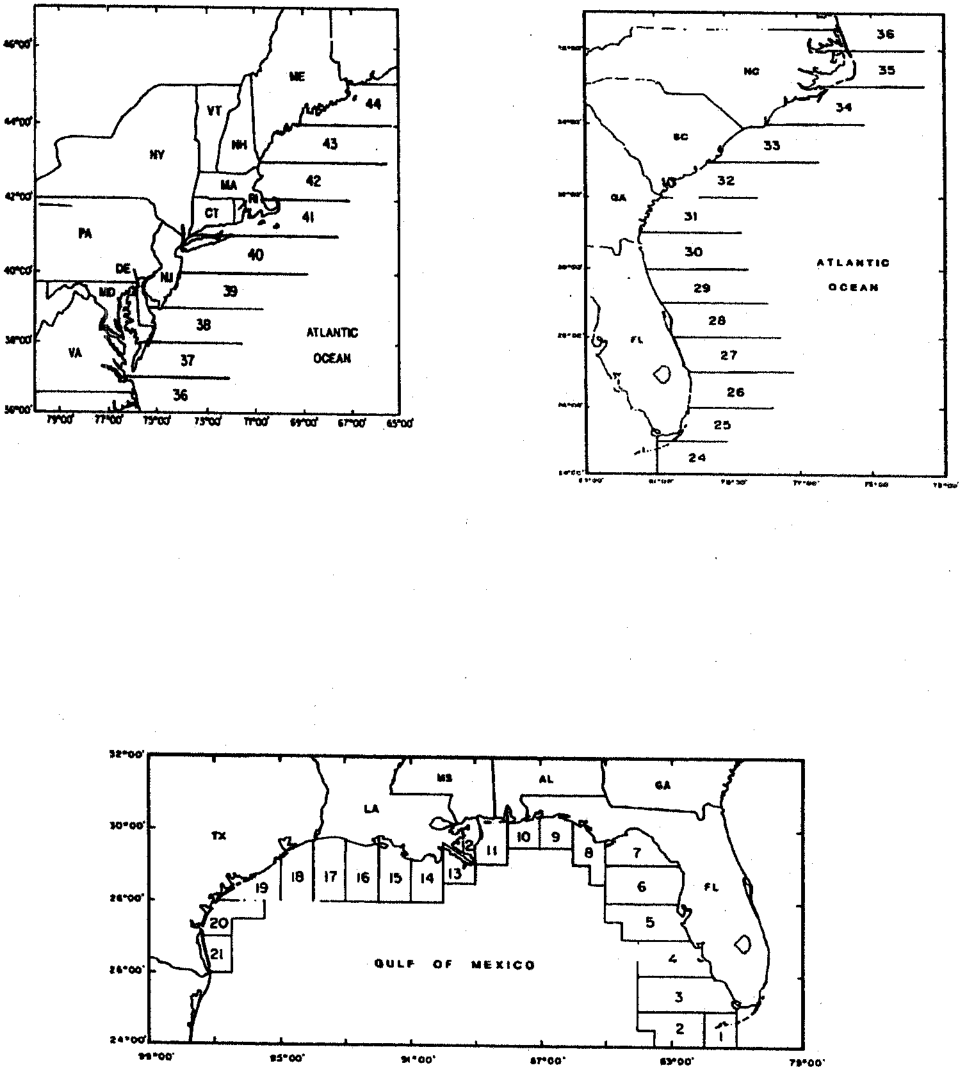
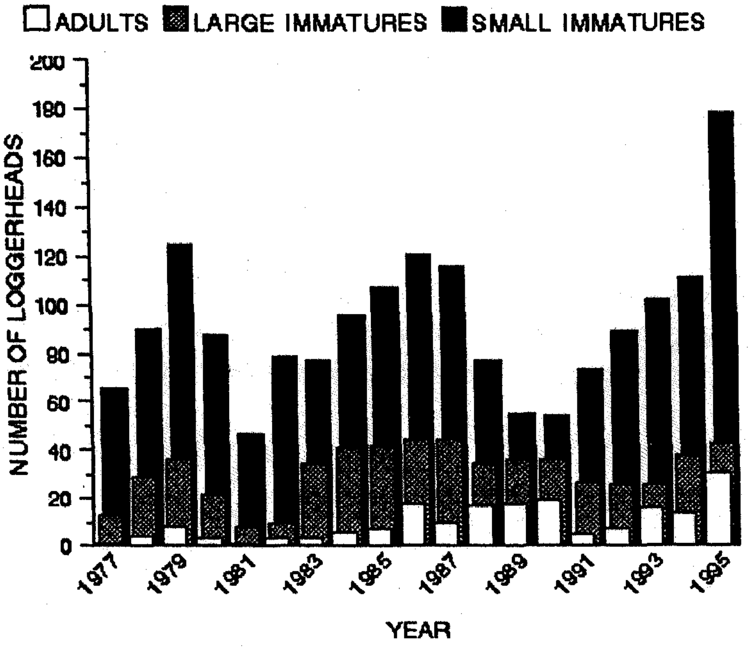


Figure 3. Captures of post-pelagic loggerhead sea turtles, 1977-1995, at the Florida Power and Light Company, St. Lucie Nuclear Power Plant. Small benthic immatures are <70 cm, large benthic immatures are 70-91 cm, and adults are ≥92 cm (Quantum Resources, 1995, unpubl. data).



APPENDIX 1

RATIO METHOD - ALL ZONES

This method estimates the minimum population size of post-pelagic loggerhead sea turtles in nearshore U.S. waters based upon the ratio of the number of stranded adult females, and nesting adult females and the number of stranded juveniles, adult males, and adults of unknown sex in a season.

The conceptual approach of this method is to use the number of nesting females during the nesting season (an accurate estimate due to extensive nesting surveys) and the number of mature females (≥92 cm) that strand along the Atlantic and Gulf coasts during the nesting season to obtain a stranding ratio. This ratio, with the number of stranded loggerheads of all size classes and both sexes on U.S. beaches in a given season can be used to estimate the total number of turtles present during that season. Adult females not present during the nesting season (with a remigration interval of 2.5 years, approximately 60% of the adult females are not present during a nesting season) are added to complete the estimate for loggerheads in U.S. nearshore waters during the period. Assumptions and details of this method are outlined below.

To estimate the number of post-pelagic loggerheads in the U.S. Atlantic population (Nt):

A. Compile nesting data 1989-94 for NC, SC, GA, and eastern FL (Table 2). Divide the number of observed nests by 4.1 nests/season (from Murphy and Hopkins, 1984) to determine the number of adult females available to strand during the nesting season (Nf).

B. Compile strandings data for zones 1-21 (Fig. 2) for the period May - August, 1989-94, by size (<92 cm and, if ≥92 cm, by sex) (Appendix Table 1). Adults (≥92cm) of unknown sex were assumed to be 1:1 males and females.

C. Solve for Nts, the total number of loggerheads available to strand (in the coastal waters) during a single nesting season, based on the proportion:

$$\frac{Sf}{St} = \frac{Nf}{Nts} \qquad \text{or } Nts = (Nf \times St) / Sf$$

where Sf is the number of stranded loggerhead adult females, St is the total number of stranded loggerheads, and Nf is the number of adult females available to strand during a single nesting season (Appendix Table 2).

D. Include the number of adult females not nesting during a given year to obtain the total number of loggerheads in the U.S. Atlantic population (Nt). Appendix Table 2 contains the results.

Nt = Nts + [(Nf x 2.5) - Nf]

Assumptions and Limitations of the Method

This method of estimating a minimum population of loggerhead turtles in the benthic nearshore waters is subject to many assumptions and limitations. These are listed below:

(1) A major limitation of this method is that it does not estimate the size of the entire loggerhead population but only the number of turtles in nearshore where strandings are believed to originate.

(2) This method assumes that strandings accurately represent the size and sex distribution of loggerheads in the nearshore waters, that the mortality causing strandings is not size or sex selective, and that the probability of a dead turtle stranding is independent of size and sex. In some years the sample size for calculating percent females in strandings was small.

(3) A limitation is that the population is estimated during the May-August time period because this is the period when nesting data are collected and, therefore, the period when the number of adult females in U.S. waters can be calculated. Additionally, sampling for strandings is the most extensive at this time of year, and has been consistent among years.

(4) The assumption that the ratio of adults of undetermined sex was 1:1 is valid only if there is no strong bias in known sex ratios. Analysis of the actual data suggested that the ratios change annually but are not particularly biased in either direction.

(5) The method assumes that subadults and males remigrate annually.

(6) The method assumes that the average number of nests/nesting female is 4.1 and that the remigration interval is 2.5 years. The effect of underestimating the number of number of clutches/female would be to overestimate the number of females in the population. The effect of underestimating the remigration interval would be to underestimate the number of females in the population.

(7) A limitation is that no nesting data were available or included in the analysis for the western Gulf of Mexico, Yucatán, or Cuba, or other Caribbean nesting beaches. No data were available for these areas and it is not known if turtles which nest in these areas

inhabit U.S. waters. The effect of excluding turtles from these areas would be to underestimate the population.

(8) No estimates of variance were available for the population estimates produced by this method. These would be expected to be large given that variance would be associated with the numerous data inputs for the method including: counts of nests, nests per female per season, remigration interval, adult female percentage in the strandings.

APPENDIX 2

RATIO METHOD - SOUTHEAST U.S. ZONES WITH SPATIAL EXPANSION

This method estimates minimum population size of post-pelagic loggerhead sea turtles in southeast U.S. nearshore waters based upon the ratio of stranded adult females and nesting adult females and the number of juveniles, adult males, and adults of unknown sex. This number is then expanded to include loggerheads in other regions of the U.S. coastal waters based on the relative abundance of turtles sighted during aerial surveys in the other regions compared to the southeast region.

The conceptual approach of this method is to use the number of nesting females located in the southeast U.S. (NC, SC, GA, eastern FL) area during the nesting season (an accurate estimate due to extensive nesting surveys) and the number of mature females (≥92 cm) that stranded in this area NMFS Statistical Zones 24-35) during a nesting season to obtain a stranding ratio. This ratio and the number of stranded loggerheads of all size classes and both sexes on U.S. beaches in a given season can be used to estimate the total number of turtles present during that season. These estimates are calculated for specific areas where substantial nesting occurs (s.e. U.S.). To obtain estimates in other areas (*e.g.*, the Gulf of Mexico and the northeast U.S.), aerial survey sighting indices are used to obtain a ratio of loggerhead abundance throughout the range. These proportions are used to calculate an estimate for the total number of loggerheads in the waters of the U.S. southeast, northeast, and Gulf of Mexico. Adult females not present during the nesting season (approximately 60%) are added to complete the estimate for loggerheads available to strand in the specific area. Assumptions and details of the method are outlined below.

To estimate the number of post-pelagic loggerheads in the U.S. Atlantic population:

I. Estimate the Number of Loggerheads (Ntse) in southeast U.S. waters

A. Compile NMFS nesting data, 1989-1994 by area: NMFS statistical zones 24-28 (s.e. FL) and zones 29-35 (n.e. FL, GA, SC, NC), corresponding to the South Florida Nesting Subpopulation, in part, and the Northern Nesting Subpopulation, respectively (Table 2). Divide the number of nests by 4.1 nests/season (from Murphy and Hopkins, 1984) to determine the number of adult females in each zone available to strand during the nesting season (Nfz).

B. Compile southeast U.S. stranding data by area (zones 24-28 and zones 29-35) for the period May - August, 1989-94 by size (<92 cm and, if ≥ 92 cm, by sex) (Appendix Table 1). Adults (≥92 cm) of unknown sex were assumed to be 1:1 males and females.

C. Solve for Ntsez, the total number of loggerheads available to strand by area (zones 24-28 and zones 29-35), based on the proportion:

$$\frac{Sfz}{Stz} = \frac{Nfz}{Ntsez} \qquad \text{or } NTsez = (Nfz * Stz) / Sfz$$

where Sfz is the number of stranded loggerhead adult females near the nesting areas; Stz is the total number of stranded loggerheads near the nesting areas; Nfz is the number of adult females available to strand in the area during a single nesting season (Appendix Table 3A,B).

Because mtDNA analyses have shown that juvenile turtles off Georgia and South Carolina are comprised of a mix of the Northern and South Florida nesting subpopulations, the number of juveniles estimated by the ratio method for Zones 29-35 was apportioned to the subpopulations in the ratio of 42% South Florida and 58% Northern (data from Sears, 1994).

D. Add Ntsez from each of the two areas to obtain Ntse.

II. Estimate the number of loggerheads in the waters of the U.S. (Nts)

A. Multiply Ntse by spatial expansion factors derived for aerial pelagic survey (from Cape Hatteras to Key West (zones 24-35) in 1982 (Thompson, 1984); Pamlico/Albermarle Sounds in 1989-91 (Epperly *et al.* 1995a), Cape Hatteras to Nova Scotia 1979-81 (Shoop and Kenney, 1992), Chesapeake Bay 1982-85 (Byles, 1988), Gulf of Mexico 1983-86 (Thompson *et al.*, 1991) to determine the number of turtles in U.S. coastal waters, but outside the southeast U.S. Atlantic (Nts) (see Appendix 3).

III. Include the number of adult females not nesting during a given year (Nfz x 2.5 years/migration) to obtain the number of post-pelagic loggerheads in U.S. coastal waters (Nt). Appendix Table 4 contains the results.

$$Nt = Nfz \times 2.5$$

Assumptions and Limitations of the Method

The assumptions and limitations for the ratio method are listed in Appendix 1, and the assumptions and limitations for the aerial expansion estimates are listed in Appendix 3. Since this method relies on both ratio and aerial data, all of those assumptions and limitations apply. This method assumes that loggerheads sea turtles are distributed as reported from the various aerial surveys. In addition, because the ratio calculations are computed by area, the problem of small sample sizes to determine the percentage of adult females in the strandings is greater than in the ratio calculation for all zones combined.

APPENDIX 3

SPATIAL EXPANSION FACTORS TO DETERMINE THE NUMBER OF LOGGERHEADS OUTSIDE SOUTHEAST U.S. WATERS

Aerial survey data were used to estimate the relative population sizes of post-pelagic loggerheads along the U.S. Atlantic and Gulf Coasts. Only surface estimates of density/numbers of turtles were used. Data for the summer months, the time period when turtles are maximally dispersed, and the time period corresponding to nesting activity in the southeast U.S. were compared.

Based on SETS surveys (Thompson, 1984, Table 10), the estimated number of loggerhead turtles on the surface in summer 1982, between Cape Hatteras and Key West, Fla., was 14,932 turtles (95% C.I.=13,978-15,886). This estimate is for offshore waters (seaward of COLREGS) only; it does not include turtles present in inshore waters. The largest inshore water body in the region is the Pamlico-Albemarle Sound, N.C. The surface estimate for all hard-shell turtles in this estuary in July 1989 was 313 turtles (95% C.I. 92-523) (Epperly *et al.*, 1995a, Table 1; density * area of strata). Epperly *et al.* (1995a, Table 6) estimated that loggerheads account for an average of 80% of the turtles in the sounds. Thus, the estimate for loggerheads alone is 250 (72-418). Note that offshore and inshore surveys were flown at 152 m. The offshore surveys were flown from a platform with downward visibility, but the inshore surveys were flown from side-viewing platforms.

Based on CETAP surveys, the estimated number of loggerhead turtles on the surface in summers 1979-1981, between Cape Hatteras and Nova Scotia, was 7,702 turtles (95% C.I.=2,220-11,000) (Shoop and Kenney, 1992, Table 3). Again, this estimate is for offshore waters, only, and does not include turtles present in inshore waters. The largest inshore water body in the region is the Chesapeake Bay. The surface estimate for all hard-shelled turtles in the lower portion of this estuary, summers 1982-1985, was 372 (95% C.I.=133-611) (Byles, 1988, Table 3). Approximately 84% of the live turtles in the Bay are loggerheads (Keinath *et al.*, 1987, Table 1). Thus, the estimate for loggerheads alone is 312 (112-513). There is not an estimate of numbers available for the upper Bay area, but it represent a smaller area and turtle density in the upper Bay during 1985-1986 was, on average, about half (47%) of that of the lower Bay (Keinath *et al.*, 1987, Table 2). Note the offshore surveys were flown at an altitude of 229 m from a platform with downward visibility and the inshore surveys were flown at 152 m from a platform with side-view visibility only.

Based on GoMex surveys in the Gulf of Mexico offshore and inshore waters and bays during the summers 1983-1986, the mean number of loggerhead turtles sighted per linear km flown in the northwest Gulf was 0.005 and the estimate for the northeast Gulf was 0.0085 (Thompson *et al.*, 1991, Table 3). If we assume the effective strip width = 0.556 km [1/f(0)=half-swath; we borrowed the f(0) calculated from other Gulf surveys using the same platform, observers, altitude, *etc.* (Lohoefener *et al.*, 1990)], then the densities in the northwest and northeast Gulf, respectively are 0.0090 turtles/sq. km and 0.0153 turtles/sq. km. Given areas, respectively, of 144,056 sq. km and 219,514 sq. km (Thompson *et al.*, 1991, p. 2), the estimated number of turtles on the surface of the Gulf of Mexico was 4,656 (1,297 in n.w. and 3,359 in n.e.). Note that these surveys were flown at 229 m altitude from a platform with downward visibility.

These data can be used to estimate crudely the relative proportion of the post-pelagic loggerhead population outside southeast U.S. Atlantic waters during the summer. Based on the above estimates of surface population sizes, by area, and given an absolute number for southeast U.S. Atlantic waters, total population sizes for each area can be obtained by multiplying the southeast Atlantic population size by these expansion factors:

Pamlico-Core Sound	0.017
Nova-Scotia to Cape Hatteras	0.516
Chesapeake Bay	0.021
Gulf of Mexico	0.312

Assumptions and Limitations of the Method

This method assumes that:

(1)The population was stable throughout U.S. Atlantic and Gulf waters 1979-1989. If this was violated, and the population was declining, the effect would be to overestimate the proportion of the population in the areas flown prior to the SETS surveys (the North Atlantic) and to underestimate the proportion in the areas flown later (Chesapeake Bay, N.C., Gulf of Mexico).

(2)The relative distribution of turtles across these areas remained constant over the time period. An independent source of the relative distribution of loggerhead sea turtles is the strandings dataset. To use these data one must assume that turtle mortality rates are equal everywhere throughout the nearshore Atlantic and Gulf of Mexico waters. The relative distribution of strandings 1991-1995 (northeast U.S. Atlantic, 21.6%; southeast U.S. Atlantic, 59.1%; eastern Gulf of Mexico, 10.1%; western Gulf of Mexico, 9.2%) is similar to the results of aerial surveys (northeast U.S. Atlantic 28.9%; southeast U.S. Atlantic, 54.5%; eastern Gulf of Mexico, 12.0%; western Gulf of Mexico, 4.6%).

(3)Either all turtles, regardless of size, are sighted, or that even if the number of small turtles are underestimated, small and large turtles are distributed similarly throughout the coastal waters.

(4)Turtles are maximally distributed during the summer and that their surfacing behavior (*e.g.* percent time on the surface) was the same in all areas during these months.

(5)All viewing platforms and altitudes flown provide equivalent results.

(6)All methods of determining density (*e.g.* strip vs line-transect methodology) yield the same results.

(7)When not specified, or data are not available to partition out loggerheads from other marine turtles sighted, all turtles sighted were loggerheads (leatherbacks were invariably separated in all studies).

In addition, there are a multitude of assumptions inherent in aerial survey methodology. See Epperly *et al.* (1995a) for a discussion of these assumptions.

APPENDIX 4

DETERMINISTIC MODEL

Frazer (1983, 1986) pieced together a life table for loggerheads based on adult females nesting on Little Cumberland Island, Georgia and mark-recapture analysis of growth and survival of benthic immatures in the southeast U.S. Atlantic. Because the number of years each turtle spends in the pelagia can be estimated from a growth curve, Frazer calculated the annual survival rate of pelagic immatures necessary for the life table to show a population decline of 3% per year. This same model fitting procedure was used in the matrix model analyses of Crouse *et al.* (1987) and Crowder *et al.* (1994). The population rate of growth or decline specified by a life table or deterministic matrix model depends on a stable age distribution, where the population has the same proportion of individuals in each age class year after year. An age-based matrix model was used by Crowder *et al.* (1994) to plot transient dynamics in nesting female population size predicted by changes in survival rates due to management. The TEWG suggested that a similar model could be used to calculate a range of possible population sizes for loggerheads from both the Northern and South Florida subpopulations. We attempted to update the published model with trends in the number of nests from both subpopulations, a new growth curve estimating age at maturity (92 cm SCL) of 20, 25 or 30 years, and new benthic immature survival rates calculated through a catch-curve analysis of strandings, 1980-1989. To get a standing stock estimate for 1989, we had to assume a stable age distribution for that year, multiplying the proportions by the total population size anticipated by the number of adult females (estimated by nest counts). Unfortunately, we ran into several problems with estimating parameter inputs. First, we had no updated estimate of adult annual survival. Because the model is driven by a stable age distribution in 1989, adult annual survival and the conversion of nests to adult females (nests/female, remigration interval) are critical. Second, it was unclear what the pre-1990 trends of each subpopulation were; while the Northern subpopulation was clearly declining in the early and mid 1980's (Hopkins-Murphy and Murphy, 1988), some Florida beaches (see Status section of report) showed an increase in nests over the same time period. Third, it was difficult to assign benthic immature survival rates and a post-1990 increase in survival (TED effect) to the subpopulations, as we do not know the genetic composition of strandings from different regions. The model structure is highly dependent on these questionable inputs, and there are no region-wide long-term data to verify the model outputs. The number of benthic immatures may be very high if the fitted parameter, pelagic immature survival, is also high, suggesting that a larger proportion of each cohort survives to recruit to the benthic feeding stage. Given these uncertainties, the TEWG decided to forego any modeling exercises for loggerheads for this report. Over the next year, attempts will be made to improve parameter estimates and produce a model that does not rely as heavily on a stable age distribution.

Appendix 4

Sources of Input Parameters for Deterministic Model

Duration of the Pelagic Stage

The best current estimate of length of the pelagic stage is 10 years (Bolten *et al.*, 1994; Zug *et al.*, 1995). Strandings data and in-water juvenile studies indicate that turtles recruit to the benthic environment between 40 and 60 cm. Although they may begin recruiting when as small as 40 cm, most begin recruiting at 50 cm (Teas, 1993; Bolten *et al.*, 1993; Epperly *et al.*, 1995b, in press; Quantum Resources, 1994). Thus, we assumed that 50 cm turtles were approximately 10 years old. Using tag recapture data, Foster (1994) predicted 51 cm turtles were 11 years old.

Size at Sexual Maturity

The size at sexual maturity was determined by examining data from Wassaw Beach, Georgia, 1991-1995, and calculating the average size of first-time nesters; this size was 92.9 cm straight carapace length (SD= 4.54, N=106) (carapace curved lengths were converted to carapace straight lengths using the equation of Teas (1993)). The average size of nesting turtles from selected rookeries of the Northern subpopulation (North Carolina: Cape Lookout National Seashore, Hammocks Beach, Onslow Beach; South Carolina: South Island; Georgia: Wassaw Island) was 93.7 cm straight carapace length (SD=5.00, N=441). There are no estimates of the size of neophyte nesters from the South Florida subpopulation for comparison. The average size of nesting turtles in the South Florida subpopulation (Merritt Island and Melbourne Beach) was 92.2 cm straight carapace length (SD=4.63, N=119). None of these sizes are significantly different from each other. Thus, 92 cm was used as the best estimate for size at sexual maturity because 1) it is within the 95% confidence interval of the mean calculated for neophyte nesters in the Northern subopulation, and 2) it does not to exceed the average size of nesters of the South Florida subpopulation.

Age at Maturity

Published literature, theses, and reports were reviewed to determine the most appropriate growth model for western Atlantic loggerhead turtles (Frazer and Ehrhart. 1985; Bjorndal and Bolten, 1988; Klinger and Musick, 1995; Frazer, 1987; Medonça, 1981; Zug *et al.*, 1986; Bolten *et al.*, 1992; Schmid, 1995; Henwood, 1987). Estimates of average age at sexual maturity varied widely (7-30 years). Sample sizes of several studies were small or included captive-reared animals; these were excluded from the review. Based on growth rates of benthic immature animals, this life history stage (50-92 cm) must require at least 10 years to complete, perhaps 20 years. Given the uncertainty in estimates of age at sexual maturity we used 20, 25 and 30 years in the model; it could be longer.

Appendix 4

Growth Curve Analysis

Von Bertalanffy growth curves were fit to two data points corresponding to the beginning of 2 major life history stages: 10 years at 50 cm, the beginning of the benthic immature stage, and 20, 25 and 30 years at 92 cm, the beginning of reproductive maturity; L infinity was constrained to equal 110 cm; a larger size is rarely observed on the nesting beaches (Ehrhart, 1979; Witherington, 1986). [A von Bertalanffy model was fit because previous studies on populations of wild immature sea turtles have shown it has a better fit than other growth equations (*e.g.* logistic, Gompertz) (Frazer and Erhart, 1985).] For an age of maturity=20 yr, a=4.966 and b=-8.305835. For an age of maturity=25 yr, a=2.4483 and b=-12.4588. For an age of maturity=30 yr, a=-0.0689 and b=-16.6117.

Catch Curve Analysis to Obtain Estimates of Survival

The von Bertalanffy models obtained from the growth curve analyses were used to convert the 1980-1995 strandings data from size distribution stranded to the age distribution stranded. The age distribution data were plotted and revealed an ascending limb and a descending limb of a curve. Ages corresponding to the younger section of the descending limb were determined and a regression of ln frequency on age for the selected age interval yielded Z, an estimate of total mortality. For age of maturity=20 yr, Z=0.3883 (SD=0..0202) over the period of 12-16 years of age (S=0.678). For age of maturity=25 yr, Z=0.2491 (SD=0.0307) over the period of 14-19 years of age (S=0.780). For age of maturity=30 yr, Z=0.2006 (SD=0.0572) over the period of 16-22 years of age (S=0.818).

Number of Adults in the Population in 1989

The mean of the estimated number of nests in the Northern, South Florida, and Florida Panhandle subpopulations for 1987-89 was divided by 4.1 (the average number of nests laid/nesting female; Murphy and Hopkins, 1984) to estimate the average annual number of females nesting each year, and multiplied by 2.5 (the average remigration interval) to estimate the total number of adult females in the population.

Hatchling/Egg Survival Rate

The model assumes that 50% of the eggs laid produce hatchlings which enter the ocean.

Primary Sex Ratio

The model was evaluated with several primary sex ratios including 3:1, 2:1, and 1:1 F:M.

Appendix Table 1. Post-pelagic loggerhead strandings, May-August 1989-1994, by zone groups, size and sex.

Number of Strandings by Zone						
Year	Size Class (Straight Carapace Length)	Zones 1-21	Zones 24-28	Zones 29-35	Zones 36-44	All Zones
1989	40.0 to 91.9 cm	88	86	275	12	461
	92.0 to 129.9 cm	17	19	25	0	61
	female	3	5	3	0	11
	male	7	6	5	0	18
	unk	7	8	17	0	32
	Total	105	105	300	12	522
1990	40.0 to 91.9 cm	74	65	289	16	444
	92.0 to 129.9 cm	14	21	35	1	71
	female	2	12	12	1	27
	male	6	1	6	0	13
	unk	6	8	17	0	31
	Total	88	86	324	17	515
1991	40.0 to 91.9 cm	57	37	155	42	291
	92.0 to 129.9 cm	18	17	28	7	70
	female	2	9	6	4	21
	male	5	4	6	0	15
	unk	11	4	16	3	34
	Total	75	54	183	49	361
1992	40.0 to 91.9 cm	47	42	194	22	305
	92.0 to 129.9 cm	11	25	31	4	71
	female	3	12	8	1	24
	male	6	4	12	0	22
	unk	2	9	11	3	25
	Total	58	67	225	26	376
1993	40.0 to 91.9 cm	50	57	188	19	314
	92.0 to 129.9 cm	21	17	29	7	74
	female	5	7	6	0	18
	male	9	7	9	4	29
	unk	7	3	14	3	27
	Total	71	74	217	26	388
1994	40.0 to 91.9 cm	121	41	250	64	476
	92.0 to 129.9 cm	19	18	53	6	96
	female	6	5	21	1	33
	male	2	5	15	2	24
	unk	11	8	17	3	39
	Total	140	59	303	70	572
Mean 89-94	40.0 to 91.9 cm	73	55	225	29	382
	92.0 to 129.9 cm	17	20	34	4	74
	female	4	8	9	1	22
	male	6	5	9	1	20
	unk	7	7	15	2	31
	Total	90	74	259	33	456

Appendix Table 2. Post-pelagic loggerhead population estimates in U.S. Atlantic and Gulf of Mexico coastal waters, 1989-1994. Estimates are based on a ratio method using May-August strandings in all statistical zones. See Appendix 1 for details on the method.

Post-pelagic Loggerhead Population Estimates

Year	Size Class (Straight Carapace Length)	Number Strandings All Zones	Percent by Size Class	Number Turtles	Offseason Females	Total All Zones	Total Zones 24-28	Total Zones 29-35	Total Zones 24-35
1989	40.0 to 91.9 cm	461	0.88314	220785		220785	118598	14785	133383
	92.0 to 129.9 cm	61	0.11686						
	female	11	0.05172	12931	19397	32328	11291	1066	12357
	male	18	0.06513	16283		16283	12546	1251	13797
	unk	32							
	Total	522		249999		269396	142435	17102	159537
1990	40.0 to 91.9 cm	444	0.86214	186198		186198	72687	15446	88132
	92.0 to 129.9 cm	71	0.13786						
	female	27	0.08252	17823	26735	44558	15139	1889	17028
	male	13	0.05534	11952		11952	4731	1336	6067
	unk	31							
	Total	515		215973		242707	92557	18671	111228
1991	40.0 to 91.9 cm	291	0.80609	138983		138983	59679	10570	70249
	92.0 to 129.9 cm	70	0.19391						
	female	21	0.10526	18149	27224	45373	15467	1646	17113
	male	15	0.08864	15283		15283	8437	1646	10083
	unk	34							
	Total	361		172416		199639	83583	13862	97444
1992	40.0 to 91.9 cm	305	0.81117	144094		144094	46597	13186	59783
	92.0 to 129.9 cm	71	0.18883						
	female	24	0.09707	17244	25866	43110	14555	1582	16137
	male	22	0.09176	16299		16299	7498	2051	9549
	unk	25							
	Total	376		177637		203503	68650	16818	85469
1993	40.0 to 91.9 cm	314	0.80928	144759		144759	89490	9260	98750
	92.0 to 129.9 cm	75	0.19072						
	female	18	0.08119	14522	21783	36305	12345	1104	13449
	male	29	0.10954	19593		19593	12345	1359	13704
	unk	27							
	Total	388		178874		200657	114180	11723	125902
1994	40.0 to 91.9 cm	476	0.83217	173119		173119	78212	9472	87684
	92.0 to 129.9 cm	96	0.16783						
	female	33	0.09178	19094	28641	47735	15663	1927	17590
	male	24	0.07605	15821		15821	15663	1535	17198
	unk	39							
	Total	572		208034		236675	109538	12934	122472
Mean 89-94	40.0 to 91.9 cm	382	0.83797	167074		167074	74735	11797	86533
	92.0 to 129.9 cm	74	0.16203						
	female	22	0.08339	16627	24941	41568	14077	1536	15612
	male	20	0.07864	15679		15679	9451	1491	10942
	unk	31							
	Total	456		199380		224321	98264	14823	113087

Appendix Table 3A. Base estimates of post-pelagic loggerhead population size from the Northern Nesting Subpopulation, before spatial expansion factors are applied. See Appendix 2 for details on the method.

Northeast FL, GA, SC, and NC

Assumes adults of unknown sex are 50:50 M:F (May-Aug Strandings)

Year	Size Class (Straight Carapace Length)	Number Strandings Zones 29-35	Percent by Size Class	Number Turtles	Percent NE FL-NC	Number NE FL-NC	Percent SE FL	Number SE FL	Offseason Females	Total NE FL-NC Turtles
1989	40.0 to 91.9 cm	275	0.91667	25491	0.58	14785	0.42	10706		14785
	92.0 to 129.9 cm	25	0.08333							
	female	3	0.03833	1066	1	1066	0	0	1599	1066
	male	5	0.04500	1251	1	1251	0	0		1251
	unk	17								
	Total	300		27809		17102		10706		17102
1990	40.0 to 91.9 cm	289	0.89198	26630	0.58	15446	0.42	11185		15446
	92.0 to 129.9 cm	35	0.10802							
	female	12	0.06327	1889	1	1889	0	0	2834	1889
	male	6	0.04475	1336	1	1336	0	0		1336
	unk	17								
	Total	324		29855		18671		11185		18671
1991	40.0 to 91.9 cm	155	0.84699	18224	0.58	10570	0.42	7654		10570
	92.0 to 129.9 cm	28	0.15301							
	female	6	0.07650	1646	1	1646	0	0	2469	1646
	male	6	0.07650	1646	1	1646	0	0		1646
	unk	16								
	Total	183		21516		13862		7654		13862
1992	40.0 to 91.9 cm	194	0.86222	22734	0.58	13186	0.42	9548		13186
	92.0 to 129.9 cm	31	0.13778							
	female	8	0.06000	1582	1	1582	0	0	2373	1582
	male	12	0.07778	2051	1	2051	0	0		2051
	unk	11								
	Total	225		26367		16818		9548		16818
1993	40.0 to 91.9 cm	188	0.86636	15966	0.58	9260	0.42	6706		9260
	92.0 to 129.9 cm	29	0.13364							
	female	6	0.05991	1104	1	1104	0	0	1656	1104
	male	9	0.07373	1359	1	1359	0	0		1359
	unk	14								
	Total	217		18428		11723		6706		11723
1994	40.0 to 91.9 cm	250	0.82508	16305	0.58	9457	0.42	6859		9457
	92.0 to 129.9 cm	53	0.17492							
	female	21	0.09736	1927	1	1927	0	0	2891	1927
	male	15	0.07756	1535	1	1535	0	0		1535
	unk	17								
	Total	303		19793		12934		6859		12934
Mean 89-94	40.0 to 91.9 cm	225	0.87049	20340	0.58	11797	0.42	8543		11797
	92.0 to 129.9 cm	34	0.12951							
	female	9	0.06572	1536	1	1536	0	0	2304	1536
	male	9	0.06379	1491	1	1491	0	0		1491
	unk	15								
	Total	259		23366		14823		8543		14823

Appendix Table 3B. Base estimates of post-pelagic loggerhead subpopulation size from the South Florida Nesting Subpopulation, in part, before spatial expansion factors are applied. See Appendix 2 for details on the method.

Florida Southeast Coast

Assumes adults of unknown sex are 50:50 M:F (May-Aug Strandings)

Year	Size Class (Straight Carapace Length)	Number Strandings Zones 24-28	Percent by Size Class	Number Turtles	Number SE FL from Zones 29-35	Offseason Females	Total SE FL Turtles
1989	40.0 to 91.9 cm	86	0.81905	107892	10706		118598
	92.0 to 129.9 cm	19	0.18095				
	female	5	0.08571	11291	0	16937	11291
	male	6	0.09524	12546	0		12546
	unk	8					
	Total	105		131728			142435
1990	40.0 to 91.9 cm	65	0.75581	61502	11185		72687
	92.0 to 129.9 cm	21	0.24419				
	female	12	0.18605	15139	0	22709	15139
	male	1	0.05814	4731	0		4731
	unk	8					
	Total	86		81372	11185		92557
1991	40.0 to 91.9 cm	37	0.68519	52025	7654		59679
	92.0 to 129.9 cm	17	0.31481				
	female	9	0.20370	15467	0	23201	15467
	male	4	0.11111	8437	0		8437
	unk	4					
	Total	54		75929	7654		83583
1992	40.0 to 91.9 cm	42	0.62687	37049	9548		46597
	92.0 to 129.9 cm	25	0.37313				
	female	12	0.24627	14555	0	21833	14555
	male	4	0.12687	7498	0		7498
	unk	9					
	Total	67		59102	9548		68650
1993	40.0 to 91.9 cm	57	0.77027	82784	6706		89490
	92.0 to 129.9 cm	17	0.22973				
	female	7	0.11486	12345	0	18518	12345
	male	7	0.11486	12345	0		12345
	unk	3					
	Total	74		107474	6706		114180
1994	40.0 to 91.9 cm	41	0.69492	71354	6859		78212
	92.0 to 129.9 cm	18	0.30508				
	female	5	0.15254	15663	0	23495	15663
	male	5	0.15254	15663	0		15663
	unk	8					
	Total	59		102680	6859		109538
Mean 89-94	40.0 to 91.9 cm	55	0.73708	65959	8776		74735
	92.0 to 129.9 cm	20	0.26292				
	female	8	0.15730	14077	0	21115	14077
	male	5	0.10562	9451	0		9451
	unk	7					
	Total	74		89487	8776		98264

Appendix Table 4. Post-pelagic loggerhead population estimates in U.S. Atlantic and Gulf of Mexico coastal waters, 1989-1994. Estimates are based on a ratio method using May- August strandings in statistical zones 24-35; NC, SC, GA and east Florida nests; and aerial survey ratios. See Appendix 2 for details on the method.

Year	Age/Sex Class	Number Turtles Zone 24-35	Offseason Females	GOM Aerial	Pam./Core Aerial	Ches. Aerial	NE Aerial	Total Z 24-35 + Aerial
1989	Benthic Immature	133383						
	Adult Female	12357	18536					
	Adult Male	13797						
	Total	159537		49746	2671	3333	82290	316113
1990	Benthic Immature	88132	Post-pelagic Loggerhead Population Estimates					
	Adult Female	17028	25542					
	Adult Male	6067						
	Total	111228		34682	1862	2324	57372	233010
1991	Benthic Immature	70249						
	Adult Female	17113	25670					
	Adult Male	10083						
	Total	97444		30385	1631	2036	50262	207428
1992	Benthic Immature	59783						
	Adult Female	16137	24206					
	Adult Male	9549						
	Total	85469		26650	1431	1786	44085	183627
1993	Benthic Immature	98750						
	Adult Female	13449	20174					
	Adult Male	13704						
	Total	125902		39258	2108	2631	64941	255014
1994	Benthic Immature	87684						
	Adult Female	17590	26385					
	Adult Male	17198						
	Total	122472		38189	2050	2559	63172	254827
Mean 89-94	Benthic Immature	86533						
	Adult Female	15612	23419					
	Adult Male	10942						
	Total	113087		35262	1893	2363	58331	234355