NOAA Technical Memorandum NMFS-SEFSC-444

ASSESSMENT UPDATE FOR THE KEMP'S RIDLEY AND LOGGERHEAD SEA TURTLE POPULATIONS IN THE WESTERN NORTH ATLANTIC

A Report of the Turtle Expert Working Group

November 2000

U. S. Department of Commerce
National Oceanic and Atmospheric Administration
National Marine Fisheries Service
Southeast Fisheries Science Center
75 Virginia Beach Drive
Miami, FL 33149

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U. S. DEPARTMENT OF COMMERCE
Norman Y. Mineta, Secretary

NATIONAL OCEANIC AND ATMOSPHERIC ADMINISTRATION
D. James Baker, Administrator

NATIONAL MARINE FISHERIES SERVICE
Penelope H. Dalton, Assistant Administrator for Fisheries

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NOTICE

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National Marine Fisheries Service
Southeast Fisheries Science Center
75 Virginia Beach Drive
Miami, FL 33149
(305) 361-4285

or

National Technical Information Service
5285 Port Royal Road
Springfield, VA 22161
(703) 605-6000
(800) 553-6847 (rush orders)

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Preface

At the behest of the National Marine Fisheries Service (NMFS), the Southeast Fisheries Science Center (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 information 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 each statistical zone that indicates 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.

Since the first meeting in June, 1995, there have been several subsequent formal meetings which have resulted in the publication of status reports for the loggerhead turtle, *Caretta caretta*, and the Kemp's ridley, *Lepidochelys kempii* (Turtle Expert Working Group. 1998. An Assessment of the Kemp's Ridley (*Lepidochelys kempii*) and Loggerhead (*Caretta caretta*) Sea Turtle Populations in the Western North Atlantic. NOAA Tech. Mem. NMFS-SEFSC-409. 96 pp.) The approaches pursued to evaluate the status and condition of these stocks are considered preliminary and exploratory and we consider this a work in progress. The current report continues with this preliminary stock assessment effort and constitutes an update to the previously published report. It is the TEWG's desire that the results published in these reports be interpreted with caution and within the extent of the assumptions both with the data and with the methodologies applied. We wish to note that the exploratory modeling 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 estimate and evaluate mortality in sea turtles.

The initial charge to the group included a request to develop a method which would establish take levels especially for the commercial shrimp trawl fishery. Actual removals would be established at levels that would not jeopardize the recovery of these species as stipulated and defined by the individual species' Recovery Plans. Determining the impacts of removal on population recovery in the absence of information on population abundance proved difficult. The TEWG pursued the use of strandings data in developing annual interim stranding limits (ISL's), recognizing that this proxy for mortality is biased low. These ISL's were estimated annually for 1996, 1997, and 1998. These levels were then monitored by NMFS on a regional basis for each of these two species. These ISL's do not constitute take levels per se and are simply a proxy for take which provides guidance relative to average strandings *v.* an unusually high incidence either over the course of a year or over a short period of time (a so-called "stranding event"). The pursuit of new methods for estimation of ISL's has been discontinued, although the methods used are provided in this and previous reports to NMFS.

It is clear to the TEWG that continued work towards developing estimates of take which do not negatively impact recovery is limited in meaning without a clear understanding of the status and condition of these stocks. It is the latter effort which will be continued. The preliminary modeling work reported here will continue to focus data collection efforts and we are confident that future assessment teams can make progress as more data become available. The intention of this report is to guide research efforts to the completion of empirical population estimates to fully evaluate the status of these stocks and evaluate their condition relative to current and future management activities and recovery goals.

Much hard work and deliberation have resulted in all of these reports to NOAA. I personally want to acknowledge all the contributions of data and hard work that were necessary to finalize this report. Considerable amounts of data were contributed that were not collected by the NMFS and these data cannot be used without the explicit permission of the originator.

Dr. Nancy Thompson, NMFS/SEFSC

Executive Summary

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 mean rate of increase in the annual number of nests has accelerated over the period 1987 to 1999. If assumptions that estimated age to sexual maturity and age specific mortality rates are correct, preliminary analyses suggest that the intermediate recovery goal of 10,000 nesting females by the year 2020 is achievable.

o It appears that the South Florida loggerhead subpopulation, which is the largest in terms of nesting females, is stable or may be increasing; the status of the Florida Panhandle and Yucatán subpopulations is unknown. Current trends for the Northern subpopulation indicate this subpopulation has declined since the 1970's and may have stabilized, but it is unlikely that the recovery goals for this subpopulation will be achieved.

o Empirical estimates of the number of turtles that can be removed from the population above natural mortality without preventing or impeding recovery cannot be made at this time with the data available. Stranding limits in the form of Interim Strandings Limits (ISL) 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 initially were derived using a risk averse approach for both species based on multi-year averages of stranding totals for each species. For the loggerhead turtles, 5-year running averages were estimated and, depending on the year or region, standard deviations were added or subtracted to this average to provide the ISL. This approach recognizes the uncertainty of the status of this species and accounts for differences in status between subpopulations. Five-year running averages were estimated for the Kemp's ridley, which were then multiplied by one half the current rate of increase observed in nests. In this way, it is recognized that the rate of increase in strandings for this species should not be greater than the population rate of increase. Additional methods used in 1998 for ISLs included Potential Biological Removal (PBR) for loggerhead adults and a method for Kemp's ridleys which attempted to match the trend in strandings with the trend in hatchling production over the past 12 years, minus 1 standard deviation for uncertainty. No method for setting strandings limits was completely satisfactory to all Group members.

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

o Estimates of mortality and growth must be developed and based on empirical data collected in the field. This includes estimates of catch and mortality associated with fisheries and other human activities throughout the range of these species. The only way to develop accurate estimates of catch and mortality in fisheries is to provide observer coverage over a statistically valid portion of the fishing effort throughout the range of these turtles.

Turtle Expert Working Group Members (as of October 1998)

Crouse, Deborah
Division of Endangered Species
U.S. Fish and Wildlife Service
4401 N. Fairfax Dr., Rm 452
Arlington, VA 22203

Crowder, Larry B.
Duke University Marine Laboratory
135 Duke Marine Laboratory Road
Beaufort, NC 28516

Epperly, Sheryan P.
Southeast Fisheries Science Center
NOAA, National Marine Fisheries Service
75 Virginia Beach Drive
Miami, FL 33149

Gabriel, Wendy L.
Northeast Fisheries Science Center
NOAA, National Marine Fisheries Service
166 Water Street
Woods Hole, MA 02543

Harris, Michael
Georgia Department of Natural Resources
2070 Highway 278
Social Circle, GA 30025

Henwood, Terry
Pascagoula Laboratory
NOAA, National Marine Fisheries Service
P.O. Drawer 1207
Pascagoula, MS 39568

Heppell, Selina S.
Duke University Marine Laboratory
135 Duke Marine Laboratory Road
Beaufort, NC 28516

present address:
Department of Fisheries and Wildlife
Oregon State University
104 Nash Hall
Corvallis, OR 97333

Márquez-Milan, Rene
Programa Nacional de Investigacion de Tortugas Marinas
Centro Regional de Investigacion Pesquera
Playa Ventanas S/N, A.P. 591
Mananillo, Colima, México, 28200

Murphy, Sally
South Carolina Department of Natural Resources
P.O. Box 12559
Charleston, SC 29422

Teas, Wendy
Southeast Fisheries Science Center
NOAA, National Marine Fisheries Service
75 Virginia Beach Drive
Miami, FL 33149

Thompson, Nancy
Southeast Fisheries Science Center
NOAA, National Marine Fisheries Service
75 Virginia Beach Drive
Miami, FL 33149

Witherington, Blair
Florida Fish and Wildlife Conservation Commission
Florida Marine Resources
Department of Environmental Protection
9700 South A1A
Melbourne Beach, FL 32951

Observers:

Conant, Therese
Office of Protected Resources
NOAA, National Marine Fisheries Service
1315 East-West Highway
Silver Spring, MD 20910

MacPherson, Sandra L.
U.S. Fish and Wildlife Service
6620 Southpoint Dr., South
Suite 310
Jacksonville, FL 32216

Possardt, Earl
U.S. Fish and Wildlife Service
University of West Georgia
Department of Biology
Carrollton, GA 30118

Schmid, Jeffrey R.
Cooperative Institute for Marine and Atmospheric Sciences
University of Miami
4600 Rickenbacker Causeway
Miami, FL 33149

Schroeder, Barbara
Office of Protected Resources
NOAA, National Marine Fisheries Service
1315 East-West Highway
Silver Spring, MD 20910

Part 1. Kemp's Ridley Status Report

Summary

The Kemp's ridley sea turtle declined precipitously from tens of thousands of nesting females in the late 1940's to around 300 nesting females in 1985. Conservation efforts by Méxican and U. S. agencies have aided this species by eliminating egg harvest, protecting eggs and hatchlings, and reducing at-sea mortality through fishing regulations. From 1985 to 1999, the number of nests observed at Rancho Nuevo, Tamaulipas, and nearby beaches has increased at a mean rate of 11.3% per year. Current totals exceed 3000 nests per year, allowing cautious optimism that the population is on its way to recovery. At current rates of increase, the population should reach 10,000 nesting females around 2020, the target given in the Kemp's ridley Recovery Plan (USFWS and NMFS, 1992). An analysis of number of nests counted versus hatchlings released suggests that there has been an increase in survivorship from hatchling to maturity in the past 5-6 years. An age-based population model was constructed to calculate potential recovery times, population size, and reasonable take limits. The model was based on growth curves of recaptured juveniles, a mortality estimate from strandings of juvenile turtles, and fitted mortality rates for other life stages. A reduction in mortality of the benthic feeding stages was required to achieve a reasonable model fit; without the mortality reduction, there was no rapid increase in the predicted number of nests in recent years. The model projections suggest that the population will continue to increase rapidly if the mortality estimates are reasonable and remain fairly constant over the next two decades. However, because of uncertainties in the mortality rates, the range of population size estimates was large (approximately 8-fold). Thus, the model was unable to adequately calculate the number of turtles in the population nor reasonable take limits. Continued effort to estimate mortality rates through mark-recapture studies may reduce the range of possible population sizes.

History of Nesting at Rancho Nuevo

Historic information indicates that tens of thousands of Kemp's ridley sea turtles nested at Rancho Nuevo, México, in the late 1940's (Hildebrand, 1963). Before then and until the late 1960's, egg harvesting occurred and nearly all eggs were removed (R. Márquez-M, *personal communication*). In 1966, a turtle camp was established in Rancho Nuevo which protected nests along approximately 25 km of the beach from April to July of each year. Over the past two decades, poaching on protected beaches has been reduced to nearly zero eggs per year. From 1966-1977, 100-400 nests per year were transplanted to a protective corral while the remaining nests were left "in situ". The "in situ" nests experienced high predation and therefore low egg survival. In 1978, a joint program between the U.S. and México was initiated which increased the length of the beach sampled and number of nests protected. The proportion of nests left *in situ* was reduced to <10%, thereby greatly increasing survival of the eggs. During the first years of the joint U.S./Méxican conservation effort (1978 – 1983), at least 10 aerial surveys of the nesting areas were completed from the U.S México border southward to Veracruz. The observation of Kemp's ridley tracks outside of the Rancho Nuevo area were few and scattered (Márquez-M, 1994). However, in recent years, more nests were observed north and south of

Rancho Nuevo and as a result the nesting beach surveys were extended in 1990 and 1991, respectively (Figure 1). The northern camp, called Tepehuajes/Ostional, covered 45-50 km of beach. In 1991, another camp was established to the south of Rancho Nuevo in Barra del Tordo, increasing total beach coverage to over 100 km. Since 1996 more small camps have been added and now over 120 km of beach is included in this protection, covering nearly all of the state of Tamaulipas to Lechuguillas, Veracruz. Because new surveys added since 1996 may inflate the population growth rate estimate, the Working Group decided to include Rancho Nuevo, North and South camps only in subsequent analyses and model runs. Analysis performed on the nests and hatchlings released from the Rancho Nuevo beach alone rarely altered the results qualitatively, and there is reason to believe that females nesting at Rancho Nuevo moved north and south following a hurricane in 1990 (R. Byles, *personal communication*).

New Nesting Information

México – Update of nest numbers 1966-1998

The nesting data presented in the previous report (TEWG, 1998) for years 1978-1995 were taken from draft reports of the U.S. Fish and Wildlife Service – Joint U.S.-México Program. We have updated this information (Table 1). The previous data were considered incomplete, as they were grouped over a season for some camps, and in some cases included data from extra camps. Some minor changes were also made to nesting data from 1966-1977 because of information provided by the Méxican Instituto Nacional de Pesca. Some discrepancies with U.S. Fish and Wildlife Service reports occur because of uncounted nests that were lost by natural predation, robbed, or left "in situ" and the exclusion of nests that were not discovered until after hatching and those that were donated to the U.S. Padre Island-Galveston experiments. The current nest tally also omits duplication of numbers of nests already included in the numbers protected in corrals or boxes. The annual number of nests and hatchlings released given in Table 1 were used in all of the trend and model analyses presented in this report.

Nesting records from South Texas

Isolated Kemp's ridley nests were documented along the south Texas coast from 1948-1994, with a total of thirteen nests being found during this forty-six year period. A primary goal of the joint U.S.-México conservation project was to re-establish a nesting colony at Padre Island National Seashore, and thousands of hatchlings were "imprinted" to that area (Woody, 1985). From 1995-1998, Kemp's ridley nesting in south Texas showed a marked increase with 4, 6, 9 and 13 nests being documented in each of these years, respectively. Of these thirty-two nests, six were confirmed by tags to have been laid by turtles which were part of the headstart experiment and were imprinted at Padre Island National Seashore, seven turtles were examined and found to be "wild" turtles, and the remaining nineteen turtles were not examined for tags. (Shaver and Caillouet, 1998).

Life Stage Definitions and Geographic Locations

Post-hatchling stage

We have defined the post hatchling stage as the time the hatchlings enter the water to age 2 years, and have assigned turtles that are 5-20 cm straight-line carapace length (SCL) to this stage. Post-hatchlings disperse from Méxican beaches of the Gulf of Mexico, presumably by major oceanic currents (Collard and Ogren, 1990), and development occurs in the epipelagia. The presence of juvenile turtles along both the Atlantic and Gulf of Mexico coasts of the U.S., where they are recruited to the coastal benthic environment, indicates that post hatchlings are distributed in both the Gulf of Mexico and Atlantic Ocean. Our stage-length of two years is an average based on evidence from three sources: 1) growth curves derived from skeletochronological analysis (Zug *et al,*. 1997) and mark-recapture data (Schmid and Witzell, 1997), 2) correlation analysis of hatchlings released and the number of strandings of small turtles (20-30 cm SCL) 1, 2, 3 and 4 years later (Table 2), and 3) preliminary data from 6 dead stranded turtles that had been tagged as hatchlings at Rancho Nuevo and were recovered 1 and 2 years after release (C. Fontaine, NMFS-Galveston, *personal communication*). Variability in growth rates among individuals likely means that the actual length of time spent in the post-hatchling pelagic stage varies from 1 - 4 or even more years.

Benthic immature stage (20-60 cm)

We have assigned turtles that are 20 to 60-cm SCL,to the benthic immature stage, representing the size class that has recruited to the coastal benthic habitat from the pelagic post-hatchling stage (Ogren, 1989). The 6 stranded turtles originally tagged on the nesting beach ranged from 22-28.5 cm SCL, suggesting that turtles of this size are fully recruited to the benthos (C. Fontaine, NMFS-Galveston, *personal communication*). We further divided this stage into small (20-50 cm SCL) and large benthic immatures (50-60 cm SCL) based on the frequency of strandings used in the mortality estimates (see modeling section below). Benthic immature turtles are found in coastal habitats of the Gulf of Mexico and the entire U.S. Atlantic coast. The location and size classes of dead turtles recovered by the Sea Turtle Stranding and Salvage Network suggests that benthic immature developmental areas occur in many areas along the U.S. coast and that these areas may change given resource quality and quantity. Growth models from mark-recapture data suggest a 7-9 year duration for the benthic immature stage (Schmid and Witzell, 1997; Appendix 1).

Adult stage

Based on size data collected on the Méxican nesting beach since the mid-1960's (Márquez-M, 1994 and unpublished data), we have defined adults as turtles larger than 60 cm SCL. The distribution of dead turtles >60 cm SCL is limited to the southeast U.S. coast and the Gulf of Mexico and are essentially absent from strandings records for the northeast U.S. coast. Recent tag returns indicate that individuals from the Atlantic tagged as immature animals do nest on Méxican Gulf of Mexico beaches and therefore do move between these water bodies (Schmid and Witzell, 1997; Witzell, 1998).

Status and Trends

Nesting data

Since 1985, the lowest nesting year (740 total nests), the number of Kemp's ridley nests counted at Rancho Nuevo, North and South Camps has increased approximately 11.3% per year (r-square = 0.94, P<0.001, 95% C.I. slope = 0.096 – 0.130; Figure 2). Increase in hatchling production from 1985-1998 was slightly less, 9.5% per year (r-square = 0.955, P<0.001, 95% C.I. slope = 0.082 – 0.108). We assume that little nesting occurred North and South of Rancho Nuevo prior to 1990 (see History of Nesting at Rancho Nuevo, above), so nests at the North and South camps are included in the totals. Like Rancho Nuevo, the number of nests counted at North and South Camps has increased since census efforts began in 1990 and 1991 (Figure 3). The annual rate of increase for nests at Rancho Nuevo only, 1985 – 1999, is 7.9% per year (r-square = 0.81, P<0.001, 95% C.I. slope = 0.056 – 0.102). It is uncertain whether the current rate of increase will continue. Also, the large number of nests counted in 1998 was unprecedented and may or may not indicate an increase in the population growth rate (preliminary data from the 1999 nesting season suggests that fewer nests were observed this year). However, we can project the number of nests with the confidence intervals predicted by the exponential trend (Figure 4). Given 2.5 nests per female, if the population continues to grow at 9.6 – 13% per year it will reach the target of 10,000 nesting females around 2014 - 2025.

Nests vs. hatchlings released 8, 10, and 12 years earlier

When we plotted the number of nests observed against the number of hatchlings released 8, 10 or 12 years earlier, we found an increase in the number of nests per cohort for the past 5-7 years, regardless of time lag (Figure 5, a-c). Heppell *et al.* (1999) found a similar result when they restricted the analysis to Rancho Nuevo nests only. Several large cohorts produced in the first years of the joint U.S.-México recovery effort do not correspond with larger nesting numbers. While a "build-up" of females is expected with increasing cohort sizes, the graphs suggest that increased hatchling production alone is not responsible for the current increase in nest numbers. An increase in hatchling-to-adult survivorship and/or adult female survival in the late 1980's to early 1990's may be responsible.

Strandings

Complete strandings information by size class, fishing zone and year are provided in Appendix 2. Figure A2.1 shows the statistical zones for which sea turtle strandings are reported. Monitoring effort is not directly comparable between zones but has been reasonably consistent over this period. Figure A2.2 a, b, and c depict the Kemp's ridley strandings reported by statistical zone, by year, 1986-1997. These data do not include incidental captures, post-hatchlings, cold-stunned, or head-started animals. Over the 12-year period 1986-97, the Gulf of

Mexico accounted for 69% of total strandings, with the southeast and the northeast accounting for 25% and 6% respectively (Table 3). Strandings were variable, but lower on average in the 1980's and have increased approximately 70% over this time frame (Table 4). The increase has been confined to the southeast and Gulf of Mexico. Strandings of Kemp's ridleys have not increased in the northeast.

Strandings "hotspots"

The western Gulf of Mexico (zones 13-21), where shrimp fishing effort is consistently high, has accounted for the highest proportion of Kemp's ridley strandings (Figure A2.2), particularly of adults. There is no survey effort in zones 15 and 16, due to inaccessibility of shoreline, and coverage is low in zones 13 and 14. The lack of data from these zones may or may not reflect a lack of strandings. Over this time frame, zones 17-21 have accounted for 44.8% of all Kemp's ridleys, including 75% of all adults, stranded over the entire east Coast and Gulf of Mexico. There appears to have been a shift in the strandings of adults over this time frame, with adults being more common in zones 18 and 19 prior to 1992, and more common in zones 20 and 21 from 1994 to the present. In 1993, in zones 13 and 14, a large group of small Kemp's stranded in late May-early June. In the eastern Gulf of Mexico (zones 1-12, partial 24-25), survey coverage is low in zones 1, 3, 6, and 7 due to inaccessibility and zone 2 has very little land mass. Zones 4 and 5 account for most of strandings in the eastern Gulf. Red tide events have also been reported periodically from these zones (Foote *et al.,* 1998).

Along the southeast U.S. coast, monitoring coverage is low in zones 24 and 25. Shrimping effort is generally high in zones 29-34. Zones 30 and 31 account for the majority of Kemp's ridley strandings in this region. In the northeastern U.S. Kemp's ridley strandings are rare, except for cold-stunning events in the winter months, which are not included in this analysis. No adult ridley strandings have been reported from this region.

Systematic surveys for sea turtle strandings do not occur in México but a few (<10), almost always adult-sized Kemp's ridleys strand each year at Rancho Nuevo during the nesting season. In 1998, 30 Kemp's strandings were recorded, but the area being monitored for nesting increased in recent years (from 25 km in the 1980's to 120 km in 1998), so it is unclear whether this represents an actual increase.

Changes in size distributions

Kemp's ridleys stranded in 1986-98 ranged from about 10-70 cm. Because the population appears to be growing rapidly, we expected to see a shift in the size distribution towards smaller turtles. We found no evidence of a shift in the mean size of Kemp's ridleys stranded over this period, but the size distribution of stranded Kemp's ridleys was significantly different in the 1996-97 compared to 1986-87 (Kolmogorov-Smirnov Test, $P<0.0001$; Figure 6). Many more large benthic immature ridleys stranded in the 1990s relative to the 1980s. While analysis of stranded turtles can potentially provide information about shifts in size distributions, and therefore population trends, strandings may not be indicative of actual population trends because of biases associated with fishing mortality.

Fishery independent, in-water studies of sea turtles have been carried out at multiple sites in the southeastern U.S. Although the objectives and target species of these studies vary, the primary goal has been to characterize regional aggregations of sea turtles. Some studies have been conducted for many years (10+), others have only recently been initiated and still others have been intermittent or aperiodic. Mark-recapture surveys of Kemp's ridleys reveal important information about size distributions and growth rates of immature turtles (Appendix 1). Ogren (1989) first suggested that aggregations of smaller Kemp’s ridley turtles were increasing in U.S. waters as a result of the protection at the nesting beach. However, no quantitative data were available to support this claim. Schmid (1998) noted a higher percentage of smaller size classes captured from 1986-95 when compared to the size distribution of Kemp’s ridleys reported in 1955 by Carr and Caldwell (1956). Furthermore, the seasonal mean carapace lengths of Kemp’s ridley turtles from the former commercial turtle fishery were 5 – 10 cm greater than those recorded in recent tagging studies. The differences may be indicative of an increased number of smaller turtles in this area or could be the result of size bias in the former turtle fishery. The mean carapace length of Kemp’s ridley turtles captured at Cape Canaveral, Florida from 1986-91 (37.0 cm, *n* = 107, Schmid, 1995) was slightly smaller than that reported from 1978-84 (38.6 cm, *n* = 40, Henwood and Ogren, 1987). However, this difference may not be significant and could be the result of sampling error owing to dissimilar sample sizes. Reynolds and Sadove (1998) reported no significant change in the mean carapace lengths of Kemp’s ridley turtles captured in New York waters from 1986-96. Despite possible temporal shifts in size classes, there are no quantitative data that conclusively demonstrate demographic shifts in the population.

In-water surveys

In-water studies targeting Kemp’s ridleys either have been or are being conducted at several sites along the U.S. Gulf of Mexico and Atlantic coasts. These sites are: Sabine Pass (Tx.), Calcasieu Pass (La.), Big Bend area (Fla.), Cedar Key (Fla.), Ten Thousand Islands (Fla..), Pamlico Sound (N.C.), Chesapeake Bay (Va.), and Long Island Sound (N.Y.). There are numerous difficulties in designing in-water studies to ensure that standardized catch per unit effort (CPUE) methods are employed such that data can be compared not only within site but also site-to-site. Environmental conditions, including weather, depth strata, current patterns, visibility, water temperature, by-catch, and investigator experience, can greatly affect turtle capture rates. Biological conditions, such as prey abundance and habitat availability, complicate sampling strategies that may be based on random distribution of the target species. Changes in effort, variability in temporal sampling, and different sampling methodologies make analyses of population trends particularly difficult.

Evaluating trends in abundance based on in-water survey data would be statistically unreliable at this time without some method of standardizing the amount of effort. Because most surveys are restricted spatially or temporally, surveys may monitor local availability at the time of the survey rather than the relative abundance of the population as a whole or a life stage of the population. High variability in survey indices due to patchy distributions means that any subsequent survey estimates will probably continue to have high variances. Trawl surveys may be a feasible means to monitor status of benthic stages of the population, but are not the only method. The Working Group recommends the design and implementation of a pilot survey, bearing in mind the patchy nature of population distribution.

The only in-water trawl surveys that appeared to have a potential for documenting long-term changes in Kemp’s ridley abundance were the NMFS data base from the 1970's and early 1980's (reported in Henwood and Stuntz, 1987) and recent trawl surveys conducted by the Gulf and South Atlantic Fisheries Development Foundation. A comparison of these data was conducted by the Foundation (1998; Jamir, 1999).

There were some differences in survey design between the two studies. The original NMFS observer program was conducted to document the rate of incidental capture and mortality for all turtle species that might encounter shrimp trawls. The survey approach was very simplistic; observers were placed aboard cooperative shrimp trawlers, the vessel fished as it normally would, and all turtle encounters were recorded. The Foundation study was conducted as a direct result of 1996 Congressional language requiring an investigation of alternatives to TEDs. The Foundation more or less attempted to repeat the original Henwood and Stuntz trawl surveys, with a primary intent of demonstrating that offshore shrimp trawlers did not take turtles and therefore, should be exempt from the TED regulations. The Foundation effort (134 sea days nearshore Gulf, 463 days offshore Gulf and 125 days Atlantic) clearly reflects the emphasis of this study, but the distribution of effort does not preclude some comparisons between the two surveys. The Foundation survey approach used observers to document take and the vessel was asked to fish as it normally would (except for 1 hour tow time restrictions in nearshore water to reduce turtle mortality).

Preliminary analysis of the two data sets indicated large increases in CPUE for Kemp’s ridley and loggerhead turtles when compared to the original Henwood and Stuntz estimates. For Kemp’s ridley turtles, as much as a 35-fold increase in CPUE was calculated for some areas. A number of factors must be considered to explain these large changes in CPUE. Most importantly, at the time of the original NMFS work, Kemp’s ridley population levels were very low, and the data were 99.9% zeros (tows where Kemp’s were not caught) with a few ones where a rare ridley was taken. Also, the fleet was not using TEDs so Kemp’s ridleys on the shrimping grounds were probably extremely rare due to high mortality rates. Thus, the CPUE calculations from the late 1970's and early 1980's were composed of rare event captures of a highly depleted stock. The Foundation study, on the other hand, was conducted after almost 10 years of full implementation of TEDs, 30 years of nesting beach protection and 10 years of increases in nests and number of hatchlings produced at Rancho Nuevo. These factors should result in an increased number of turtles available for capture, and the Foundation data supported these expectations. Foundation observers were able to catch more turtles with much less effort which resulted in huge changes in CPUE.

While the observed increases in CPUE are impressive, caution should be exercised in interpreting these results. When dealing with small CPUE values (as with the old NMFS estimates Kemp's ridley), any increase in number of turtles caught at a given level of effort or decrease in effort with the same number of turtles caught can result in some "multi-fold" increase in CPUE estimates. The Foundation study indicates that there are now Kemp's ridleys in nearshore waters in measurable quantities, and these turtles were not there or were present in very low abundance 15 years ago. What CPUE will not explain in this case is how much the total population has increased. To illustrate this point consider a comparison of CPUE from an overfished stock where 95% of the fish have been taken from the habitat, with CPUE from that same habitat after ten years of protection has allowed 100% reoccupation of that habitat. Now consider that this habitat is only one of many occupied by this species, the rest of which have never been fished. A twenty-fold increase may have occurred in the overfished habitat, but that tells us nothing about the total population increase.

The Foundation catch rates of sea turtles indicate that future in-water trawl surveys could produce enough captures to allow monitoring of population abundance and trends in a cost effective manner. If such surveys were conducted, a randomized sampling design should be used to cover the entire coastline and not just the shrimping grounds where active fishing is occurring. This is an important consideration to note because while the NMFS and Foundation data sets allow estimation of CPUE on the shrimping grounds, nothing can be said about turtle abundance in areas or times when shrimp fishing is not occurring.

Trawl surveys are probably the best currently available means of obtaining information on the in-water abundance of sea turtles. The Foundation study, while suffering from less than optimal effort allocation, provides some good insights regarding the effectiveness of TED regulations. The increases in CPUE observed in this study are the strongest evidence to date that Kemp's ridleys are coming back on the shrimping grounds where they were almost eradicated two decades ago. The CPUE values reported are encouraging and believable, although they do not imply multi-fold increases in the total population. What they do suggest is multi-fold increases in areas where the species was severely depleted and kept down as a result of shrimp trawling pressure. Kemp's ridleys now appear to be repopulating their former habitat.

Conclusions from nesting and strandings data

There is evidence from nest numbers and in-water surveys that the Kemp's ridley population is increasing. There has apparently been an increase in survivorship to maturity and the proportion of large benthic immatures in the strandings records has increased, suggesting that juvenile mortality has decreased. The number of strandings in the Gulf of Mexico increased sharply in 1994 and 1995 but remained fairly constant (about 450 turtles per year) thereafter. It is reasonable to believe that some of the recent increase in strandings is due to an increase in population size. However, because of the time lag required for each cohort to reach maturity and recruit to the nesting population, there is not a clear connection between changes in strandings or in-water surveys and changes in the number of nests, nor is there an easy way to estimate the effect of changes in juvenile fishing mortality on the nesting female population. We attempted to elucidate this relationship using a population model, described below.

Nest protection in México – Future strategies

Since 1978, nearly all Kemp's ridley nests have been moved to protected "corrals" for incubation, thereby greatly reducing the loss of eggs and hatchlings to erosion and predation. As the population continues to increase and expand north and south, the proportion of nests that can be completely protected will decrease. From 1987 to 1999, the number of nests reported has increased annually from less than 750 to approximately 3400 for Rancho Nuevo, Tepehuajes and Barra del Tordo camps. Based on the relative importance of these camps for nesting and the number of workers and equipment that can be accommodated, it is estimated that the total number of nests that can be protected in corrals is about 5,000. This 5,000 total includes about 4,000 from Rancho Nuevo and about another 1,000 at the north and south camps. Another 1,000 nests could be protected "in situ" with wire screens. Any remaining nests will have to remain where laid with increased protection through increased beach monitoring with additional beach workers. In addition, the number of beach workers required each year is determined by the nesting behavior of the turtles. In some years, the turtles aggregate along specific areas of the beach and fewer beach monitors are required as compared to years when nests are scattered. While efforts will be made to protect as many nests as possible, it is realistic to anticipate that the current nest survival rate of approximately 65% will not continue indefinitely.

Population size and trends from a deterministic age-based model

Model modifications from the 1998 report

Models are most useful when they are updated with new data and modified according to the best information available. The demography of sea turtles is, generally, poorly understood, and new information must be incorporated into model form and parameterization whenever possible. Changes to the model that we developed in 1995 are the results of new information and discussion:

1) The small benthic immature mortality rate is based on a re-analysis of strandings information, using a variety of cohort-based approaches to account for changes in cohort size and variance in strandings levels, rather than pooling year classes. This mortality rate is applied to age classes 2 - 6, based on the growth curve from mark-recapture data (Appendix 1).

2) The mortality rate of large benthic immatures is now equal to that of adults, rather than small benthic immatures, based on the poor fit of the catch curves to strandings of age classes greater than 6 years and the argument that large immature turtles (>50 cm SCL) are likely to have a lower mortality rate (and more similar to the adult mortality rate) than smaller turtles.

3) The number of nests used in the model fitting exercise has been updated according to Table 1.

4) The change in nest survival with nest density has been updated according to recent discussions with the INP and camp managers.

5) The number of years used to calculate the fitted parameters (pelagic immature and adult survival and the post-1990 mortality decrease) has been varied to achieve the model that best-fits both past and recent trends in the nesting data.

These changes have altered the fitted parameters, expected growth rates, and population size estimates of the models. We have done a sensitivity analysis similar to that shown in the 1998 report. These models represent our understanding to date of Kemp's ridley population dynamics, but will undoubtedly change as more information becomes available to run more complex simulations.

Model form

A deterministic age-based model was implemented with the short-term objective of providing a framework to organize and evaluate available data and parameter estimates. The development of a model to track stock status in terms of abundance and mortality at age, to project stock abundance levels conditional on current stock status, and to provide management advice will require reliable estimates of input parameters which are currently unavailable.

The number of female hatchlings produced in each year (Table 1) are projected forward as a cohort. Each cohort passes through four different life stages (Table 5), where age at sexual maturity (is determined by a growth curve (Appendix 1). All animals aged or higher are assumed to undergo mortality at the rate of mature individuals and reproduce every other year.

The model is used to estimate three parameters through a least-squares analysis that compares nests predicted by the model to nests observed each year (Table 1). Instantaneous total mortality is estimated for two age groups: pelagic immature (Z_p), and the combined large benthic immature-adult stages (Z_a). A mortality rate for small benthic immature turtles (Z_b) is based on empirical data (strandings, see below) independent of the model. The mortality rate of the large benthic immature stage was assumed to be more similar to that of the adult stage than to that of the small benthic immature stage. This is because large benthic immature animals are rare in strandings in the Gulf of Mexico and are assumed to be distributed in different areas from small benthic immature animals. This makes the large benthic juveniles potentially less vulnerable to mortality than small benthic immature animals. A third parameter reflecting a post-1990 decrease in mortality (f) was also estimated. This multiplier was applied to the small benthic immature and large benthic immature-adult mortality rates after 1990 to address a lack of fit (a trend from maximum negative residual values in 1989 to maximum positive value in 1998) which arises when a single mortality rate per life history phase is applied over the duration of the 1966-1999 time series (Figure 7, in Model Results below). The Excel 7.0 Solver algorithm was used to estimate the parameters by comparing predicted numbers of nests to observed values.

Thus, five different mortality rates are obtained over the time and age range of the model (Table 6) where Z_i, Z_a, and f are estimated using the model fitting exercise and Z_b is input or estimated independently. The post-1990 multiplier (f) is a number less than 1.0, representing a proportional decrease in Z_b and Z_a. Mortality rates are not re-estimated annually. Observational data consist of annual numbers of nests at Rancho Nuevo and north and south camps (1966-1999) and annual hatchling production (1996-1998), which serves as cohort size for each year (Table 1). Input parameters included adult female remigration rate, mean number of eggs per nest, female age at maturity, sex ratio in the population, and survivorship of benthic immature turtles (see below). The temporal resolution is annual, with time periods beginning 1 July, corresponding with average time of hatchling production and nesting. The number of adults is modeled without age structure and no maximum age is set.

Starting population sizes were derived from estimates of nests and hatchlings in 1966 (Table 1). Past analyses have shown that the model is relatively insensitive to the nest estimate used in 1966, although this value does affect the model estimate of adult survival (Appendix 2 and 3; TEWG 1998). 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, and consequently that there was no recruitment to the nesting population in 1966-1976.

The number of female hatchlings is based on an assumption of a 1:1 sex ratio for hatchlings produced, and of no other surviving hatchlings from sources other than monitored nesting beaches at Rancho Nuevo, North and South Camps. The ratio between hatchlings produced and subsequent contribution of a cohort to the nesting population is assumed to be constant for all cohorts recruiting by 1990, because age-specific mortality rates are fixed for all ages over that period, after which the effect of the mortality multiplier is incorporated. Similarly, reproductive parameters, including remigration rates, number of nests per female, and growth rates (*i.e.*, stage lengths and age at maturity) are assumed constant over the 1966-1999 series.

Parameterization

Nesting data used

We considered the modeled population to include nesting and hatchling production at Rancho Nuevo as well as the North and South Camps. Prior to the mid 1980s, aerial surveys documented few crawls outside Rancho Nuevo; it seems unlikely that many hatchlings were produced given the small numbers of nests and that there was no protection afforded to the eggs or hatchlings. So we assume that hatchlings estimated at Rancho Nuevo led to nests subsequently detected at Rancho Nuevo and elsewhere. Given that Kemp’s ridleys nest numbers reached their lowest numbers in the mid-1980s, one might expect their range to expand with increases in the number of females in the nesting stage. Nesting surveys began at North and South Camp in 1990 (Table 1).

In the 1998 report, the model was fit to years 1978-1995, representing all years for which there were complete nest counts. Nesting information is now available through 1999 (Table 1). We wanted to produce a model that fit the time series well but could also be verified by recent nest counts and used to roughly estimate future population growth. The number of nests observed in recent years is more variable than in the 1980's (Figure 2), and the number of years used to fit the model can affect the fitted survival rates, the post-1990 multiplier, and the model projections. We ran a sensitivity analysis to see how the addition of each new year of data would affect the model fit for the model fit years (1978-199x), early years (1978-1995), recent years (1996 to 1999) and for the entire available time series (1978-1999) (Table 8). We used $Z_b = 0.5$ for the analysis (Z_b does not affect the residuals; TEWG, 1998).

Reproduction

For the model fitting exercise, nests deposited per female and years between nestings (remigration interval) were needed to convert nests to adult females, and sex ratio was needed to convert females to total population size. Cohort sizes for years 1966 - 1998 were the number of hatchlings released from Rancho Nuevo, North and South camps, so additional reproductive parameters, such as eggs per nest and hatch success, were not needed until we projected population size into future years. For model projections, we examined the effects of reduced nest survival with nest density, as might be expected as more nests are left *in situ* (Table 7). We considered published and unpublished information in developing the following assumptions and estimates for reproductive parameters:

1. The **number of nests per female per season** was calculated as 3.075 in a physiological and ultrasonographic study by Rostal (1991). Pritchard (1990) estimated 2.31 nests per female per season, and a recent tagging study using PIT (passive integrated transponder) tags (R. Márquez-M., *personal communication*) estimated 1.8 nests per female per season. We used **2.5 nests per female per season**, which is close to the mean of means (2.4 nests per female per season) of these studies.

2. **Adult female remigration rate** to the nesting beach was estimated from mark-recapture data (Márquez-M *et al.* 1989). We updated these data with current information (R. Márquez-M., *personal communication*). Roughly 20% of the turtles nest annually, 60% nest every 2 years, 15% nest every 3 years, and 5% every 4 years. The weighted mean remigration rate was 2.05 years, which we truncated to **2 years** and used in the model. However, we recognized that estimation of remigration rate must take into account adult mortality rate (Frazer 1989) and the probability of encountering nesting females (Pritchard, 1990), but neither of these were known.

3. A **mean number of eggs per nest** of 100.8 was noted by U.S. Fish and Wildlife Service and National Marine Fisheries Service (1992), which we truncated to **100 eggs per nest**.

4. For population projections beyond 1999, **hatch success** (*i.e.*, egg survival) for nests in corrals was set at 65%, based on the mean of [annual hatchlings released/(nests*100 eggs/nest)] for years 1990-1998 (Table 1). **Hatch success** was reduced step-wise to simulate a reduction in nest protection with nest density. Protection of nests from predators and poachers at Rancho Nuevo nesting beaches is effort-dependent, and can change with time due to changes in numbers of nests laid and availability of resources to protect them. Under previous plans of the Instituto Nacional de la Pesca (INP) and U.S. Fish and Wildlife Service (USFWS), an agreed upon maximum of 5,000 nests would be protected in corrals, with a maximum of an additional 1,000 nests protected *in situ* by screening. Nests in excess of 6,000 would be left in natural surroundings with no protection, except perhaps predator control and beach patrols. For our model projections, the hatch rates were imposed on the model according to such best approximation protection scenarios (Table 7).

Sex ratio (all age classes)

For the model exercise in our first report we assumed a sex ratio of 1:1 (TEWG, 1998). Since then two studies in the northern Gulf of Mexico have provided support for that assumption (Coyne and Landry, 2000; Gregory and Schmid, *in review*). Although no significant differences were found from a 1:1 ratio across months and sampling locations, females did comprise the majority of turtles sampled in these areas.

Age and growth (see Appendix 1)

Age at maturity for Kemp's ridley turtles was estimated by Márquez-M (1972) as 5-6 years. The U.S. Fish and Wildlife Service and National Marine Fisheries Service (1992) thought that estimate too low, but offered no better estimate. Zug *et al*. (1997) estimated Kemp's ridley maturity between 11 and 16 based on skeletochronological age estimates for stranded turtles and an assumed mature size of 65 cm SCL. Caillouet *et al*. (1995) developed a growth model for known-age head-started Kemp's ridleys and estimated a 10 year age at maturity, assuming size at maturity of 60 cm SCL. Schmid and Witzell (1997) estimated age to maturity at 8 – 13 years based on the recapture data of wild subadults in Florida and the minimum and mean carapace lengths of nesting females. An analysis of the NMFS Miami Laboratory tagging database (Appendix 1) indicates an 8-12 year age to maturity estimate using the criteria of Schmid and Witzell. Given the variability in estimates, we assumed the actual age at maturity for Kemp's ridley turtles to be within 7 - 15 years. Because size at first nesting and growth rates vary, we ran preliminary model runs with 8, 10 and 12 years to maturity to test the model's sensitivity to this parameter. We assumed "knife-edge" maturity (all individuals become mature at the given age) because there is no information available on the proportion mature at age or size.

Benthic immature mortality from strandings catch curve estimates (Z_b)

Catch curves were constructed both by cohort and by standing stock based on length composition of strandings data and updated growth curves. Strandings at length by month and year were re-indexed for consistency with aging by cohort slicing: the "Ridley year" *i* was defined from 1 July, calendar year i to 30 June calendar year *i*+1. An updated von Bertalanffy growth curve (Appendix 1) was used to assign age based on length, assuming a 1 July birthday. Year class was assigned by subtracting (integer) age from Ridley year.

To generate catch curves based on cohorts, observations were weighted to adjust for the potential interannual variability in strandings based on variable effort. A weighting factor for each year was calculated as the ratio of strandings in year *x* /strandings in 1991, the year of the lowest strandings, and each observation was transformed by dividing by this weighting factor. Thus, one individual stranding in year 1991 would be reflected as a 1, while an individual stranding in other years would be reflected as a number less than 1, with larger reductions in observations from years with high total strandings.

Weighted observations were then categorized by age and Ridley year, natural log transformed (ln), and a catch curve fitted for each cohort. The catch curve began with the age corresponding to the highest left-most point in the ln(frequency)-age plot, and continued to include the oldest age before an inflection point was detected by eye. For most plots, the regression line fit ages 2-6. Older age classes had low and variable catches, so we decided to apply the calculated mortality rate to ages 2-6 only and apply the adult mortality rate (Z_a) to immature turtles aged 7 - (a-1).

A similar set of catch curve analyses was generated using standing stock year, with weighting by hatchling production for a cohort, instead of weighting by strandings levels by standing stock year. These analyses gave qualitatively similar results but were considered less reliable because of variance in strandings levels. Overall, the range of Z_b was 0.3 to 0.8, with a mode of 0.5.

As outlined in the Model Form section above, pelagic immature and large benthic immature and adult instantaneous mortality rates (Z_p and Z_a) were obtained through least-squares fit of the number of nests predicted by the model and observed nests. In general, pelagic immature mortality correlated negatively with benthic immature mortality to achieve the "correct" number of nesting females (see sensitivity analysis below).

Model Results

Model fits to observed nests

We considered 4 factors in the model fitting exercise:
1) age at first reproduction
2) which nesting years to include
3) small benthic immature mortality
4) presence or absence of the post-1990 multiplier.

The lowest sums-of-squares for 1978-1999 were naturally for models fit to the entire time series, but often these models had poor fits for earlier years (Table 8). The lowest sum-of-squared residuals for models fit to fewer years were: 1978-1998 for the 8 years to maturity model, 1978-1996 for the 10 years to maturity model, and 1978-1995 for the 12 years to maturity model. We used the mortality rates fitted for these years for the model projections. The number of years used to fit the model primarily affected the post-1990 multiplier, *f*, which ranged from 0.4 to 0.7, and the model's future exponential rate of increase, which ranged from 12-17% per year. Without the post-1990 reduction in mortality for benthic immature and adult turtles, the model cannot fit the observed trend in nests, regardless of age at maturity or the number of years used to fit the model (Figure 7, example is for 10 years to maturity, $Z_b = 0.5$, model fit to nests observed from 1978-1996).

For simplicity, Figures 8(a, b, and c) show only the best fits and residuals for models with 8, 10 or 12 years to maturity, respectively, and small immature $Z_b = 0.5$. All models shown include the post-1990 multiplier. The best fit parameters for pelagic immature mortality (Z_p), large immature/adult mortality (Z_a), and the post-1990 multiplier are different for each age at maturity (Table 9). Recall that the fitting algorithm chooses these three parameters to minimize the sum-of-squares difference between the model nest count and the observed nest count. In order to achieve the rapid rate of increase observed in nest counts over the past decade, the age 12 model requires a lower post-1990 multiplier, corresponding to a more drastic reduction (50%) in the mortality rate of benthic feeding juveniles and adults. The resulting survival rates and, in turn, the population growth rate after 1990, are higher in the age 12 to maturity model (see Best-fit Model Projections).

Sensitivity of model results to changes in small benthic immature Z_b

Changes in small benthic immature mortality have virtually no effect on the model fits, because increases or decreases in one mortality rate are compensated for by the others. To illustrate this, we plotted the pelagic immature, small benthic immature and large benthic immature instantaneous mortality rates given by the best fit models for small benthic immature Z_b (pre-1990) ranging from 0.3 to 0.8 (Figure 9). While most of the compensation occurs as a trade-off between mortality in the pelagic phase and mortality in the subsequent small benthic immature stage, there is also an increase in the post-1990 multiplier with increasing small benthic immature mortality (Figure 10). Recall that the multiplier is a proportional change in mortality, so higher multipliers result in smaller changes in annual survival pre- and post-1990. The mortality compensation results in a very similar predicted number of nests, but very different population size estimates (Figure 11). Because we are trying to fit three parameters with the current model, there are a large number of feasible solutions to the mortality schedule. Thus, until empirical estimates can be derived for the post-1990 multiplier and pelagic immature, large benthic immature and adult mortality, it would be imprudent to use this model to estimate population size.

Model fits with an empirically-derived estimate of *f*

We estimated the post-1990 change in small benthic immature mortality by comparing the strandings catch curves for cohorts that stranded before and after 1990. If pre- and post-1990 estimates of Z_b are included in the model, (Z_b = 0.52 and 0.34, respectively), the fit and model outputs are similar to the small benthic immature mortality = 0.5 run with 10 years to maturity (Figure 12 a). The model predicted a greater post-1990 reduction in large immature and adult mortality (f = 0.52) than the reduction estimated empirically by the catch curve coefficients of small benthic immatures (0.34/0.52 = 0.65). It is not known if this is the result of imprecision of Z_b estimates or if there is a larger post-1990 mortality decrease for larger size classes of turtles. When the post-1990 multiplier was fixed at 0.65, the model had a lower fit and biased residuals (sum-of-squared nest residuals 1978-1999 = 791,064; Figure 12 b).

Best-fit model projections

In spite of the large differences in total population size, the model runs with small benthic immature Z_b ranging from 0.3 to 0.8 showed a relatively small range of predicted population growth rates, measured as annual proportional increase and assuming that mortality rates are constant through time (range 12 – 16% per year). In general, models with lower Z_b (= higher annual survival for 2-6 year-olds) predicted slower population growth rates (these models also had less reduction in mortality after 1990). Because of their similar post-1990 survival rates, the age 8 at maturity and age 10 at maturity models showed similar population trajectories (Figure 13). The age 12 at maturity model, with its higher post-1990 survival rate estimates, predicted a faster growth rate despite the later age at first reproduction. The time required to reach 10,000 nesting females varied somewhat in the sensitivity analysis (range 12 – 16 years for the age at maturity = 10 years model). A reduction in hatch success as the nest population increased greatly reduced the expected population growth rate for all of the models, but not until after the 10,000 nesting female mark was reached (example shown is 10 years to maturity, pre-1990 small benthic immature Z_b = 0.5; Figure 14). These model projections are for comparative purposes only - they are not meant to forecast population size through time. The models do not include environmental variance and we do not know which, if any, of our models most accurately matches the mortality schedule and life cycle of the Kemp's ridley. As more information on survival and growth is obtained from field studies, we will be able to reduce the number of unknowns and produce more predictive models.

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Table 1. Nests counted at Rancho Nuevo, North Camp and South Camp 1966 - 1998. Nests and hatchlings released are approximate for 1966 - 1977. Total nests includes estimates (1966 – 1977) or counts (1978 – 1998) of nests not moved to protective corrals. Total hatchlings = hatchlings released from corrals only. **Hatchling count not yet determined for 1999.

Year	Rancho Nuevo	North Camp	South Camp	Total Nests	Total Hatchlings Released	Hatchlings Released per Nest
1966	5991			5991	29100	4.9
1967	5519			5519	24100	4.4
1968	5117			5117	15000	2.9
1969	4018			4018	28400	7.1
1970	3017			3017	31400	10.4
1971	2012			2012	13100	6.5
1972	1824			1824	14600	8.0
1973	1642			1642	23500	14.3
1974	1466			1466	23500	16.0
1975	1266			1266	11100	8.8
1976	1100			1100	36100	32.8
1977	1036			1036	30100	29.1
1978	1012			1012	48009	47.4
1979	1019			1019	63996	62.8
1980	900			900	37378	41.5
1981	962			962	53282	55.4
1982	842			842	48007	57.0
1983	795			795	32921	41.4
1984	861			861	58124	67.5
1985	740			740	51033	69.0
1986	752			752	48818	64.9
1987	742			742	44634	60.2
1988	840	0	3	843	62218	73.8
1989	905	11	0	916	66802	72.9
1990	800	204	0	1004	74339	74.0
1991	862	178	74	1114	79749	71.6
1992	932	254	84	1270	92116	72.5
1993	847	251	101	1199	84605	70.6
1994	1126	329	110	1565	107687	68.8
1995	1438	334	149	1921	120038	62.5
1996	1255	444	219	1918	114842	59.9
1997	1514	358	347	2219	141770	63.9
1998	2409	431	642	3482	167105	48.0
1999	2308	273	776	3357	**	**

Table 2. Small turtle strandings (20-30 cm SCL) and hatchlings released 1 - 4 years prior to stranding year, 1986–1997.

		Hatchlings released			
Year	Strandings	1 year earlier	2 years earlier	3 years	4 years
1986	105	51033	58124	32921	48007
1987	30	48818	51033	58124	32921
1988	24	44634	48818	51033	58124
1989	27	62218	44634	48818	51033
1990	28	66802	62218	44634	48818
1991	25	74339	66802	62218	44634
1992	24	79749	74339	66802	62218
1993	94	92116	79749	74339	66802
1994	121	84605	92116	79749	74339
1995	40	107687	84605	92116	79749
1996	41	120038	107687	84605	92116
1997	38	114842	120038	107687	84605
Correlation		0.2603	0.4047	0.1419	0.2833

Table 3. Kemp's ridley strandings, 1986-1997.

	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	Total
N.E. U.S.	27	12	24	10	11	12	20	25	25	16	14	16	212
S.E. U.S.	32	51	116	43	94	48	30	79	95	91	114	92	885
Gulf of Mexico	246	98	75	96	153	105	106	223	430	288	255	304	2379
Total	305	161	215	149	258	165	156	327	550	395	383	412	3476

Table 4. Kemp's ridley strandings trends. Percent of total strandings represented by each location is shown in parentheses.

	1986-88 total	1995-97 total	Percent change
N.E. U.S.	63 (9%)	46 (4%)	-27%
S.E. U.S.	199 (29%)	297 (25%)	+49%
Gulf of Mexico	419 (62%)	847 (71%)	+102%
Total	681	1190	0.75

Table 5. Life stages defined in the age-based model for Kemp's ridleys.

Life stage	Component ages
Pelagic immature	0 through 1
Small benthic immature	2 through 6
Large benthic immature	7 to (-1) (*e.g.*, 9 if 10)
Adult	+ (*e.g.*,10+)

Table 6. Mortality rate parameters defined for the age-based model.

Stage	Pre- 1990	Post-1990
Pelagic immature	Z_p	Z_p
Small benthic immature (age 2-6)	Z_b	$f{*}Z_b$
Large benthic immature	Z_a	$f{*}Z_a$
Adult	Z_a	$f{*}Z_a$

Table 7. Hypothetical changes in hatching rate expected with increasing numbers of nests, as applied to density-dependent projections of the population model.

Range in nests	Protection regime	% survival to hatchling release
0-5,000	Corral	65
5,001-6,000	in situ with screening	50
6,001-10,000	in situ with predator control	30
>10,000	in situ with no predator control	20

Table 8. Sum of squared residuals (observed - expected nests) for 3 models fit to various time series. Small benthic immature $Z_b = 0.5$.

8 years to maturity				
fit to:	model fit yrs (1978-199x)	1978-1995	1996-1999	1978-1999
1995	126,679	126,670	2,149,482	2,276,151
1996	209,839	185,056	429,312	614,368
1997	235,948	221,965	712,188	934,153
1998	526,370	166,135	428,933	595,069
1999	561,440	171,175	390,264	561,440
10 years to maturity				
fit to:	model fit yrs (1978-199x)	1978-1995	1996-1999	1978-1999
1995	60,823	60,823	1,354,351	1,415,174
1996	86,669	77,980	360,795	438,776
1997	89,642	85,029	435,837	520,867
1998	214,790	79,748	742,805	822,554
1999	429,941	73,829	356,111	429,941
12 years to maturity				
fit to:	model fit yrs (1978-199x)	1978-1995	1996-1999	1978-1999
1995	161,174	161,174	589,762	750,937
1996	165,800	163,847	923,858	1,087,706
1997	166,084	164,770	997,803	1,162,573
1998	447,024	208,158	720,433	928,592
1999	578,015	165,934	412,081	578,015

Table 9. Parameters for model best-fits with small benthic immature z=0.5.

Age at maturity	Pelagic immature annual survival		Large benthic immature and adult annual survival (age 7+)		Post-1990 multiplier (*f*)
	pre-1990	post-1990	pre-1990	post-1990	
8	0.24	0.24	0.85	0.91	0.56
12	0.37	0.37	0.86	0.92	0.50
10	0.31	0.31	0.85	0.91	0.56
10 - w/out post-1990 multiplier	0.38	0.38	0.83	0.83	1.0

Figure 1. Nesting beaches of the Kemp's ridley sea turtle. Primary nesting beach is Rancho Nuevo

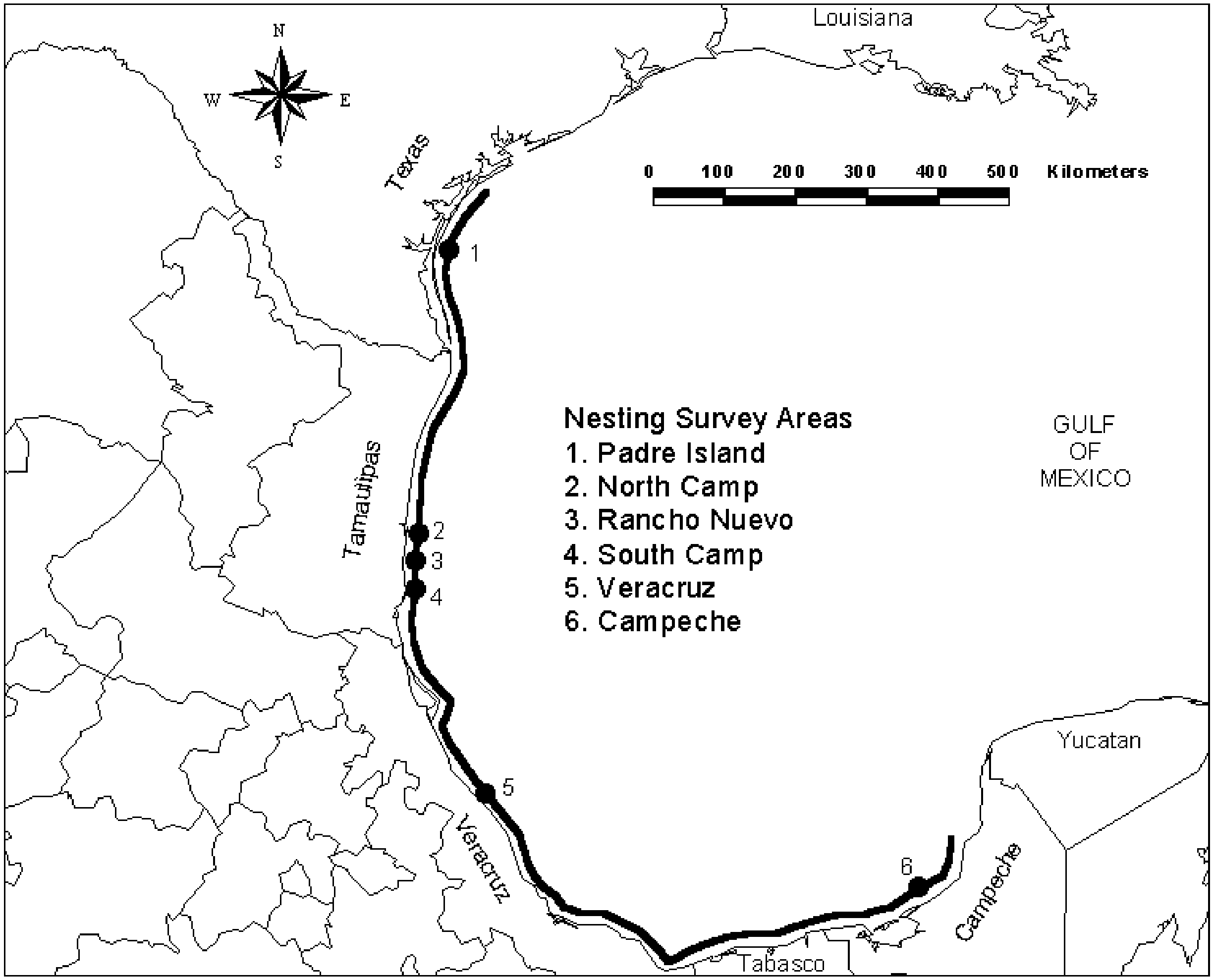


Figure 2. Nests and hatchlings released from Rancho Nuevo, North Camp and South Camp, 1966-1999 (hatchlings for 1999 are approximate).

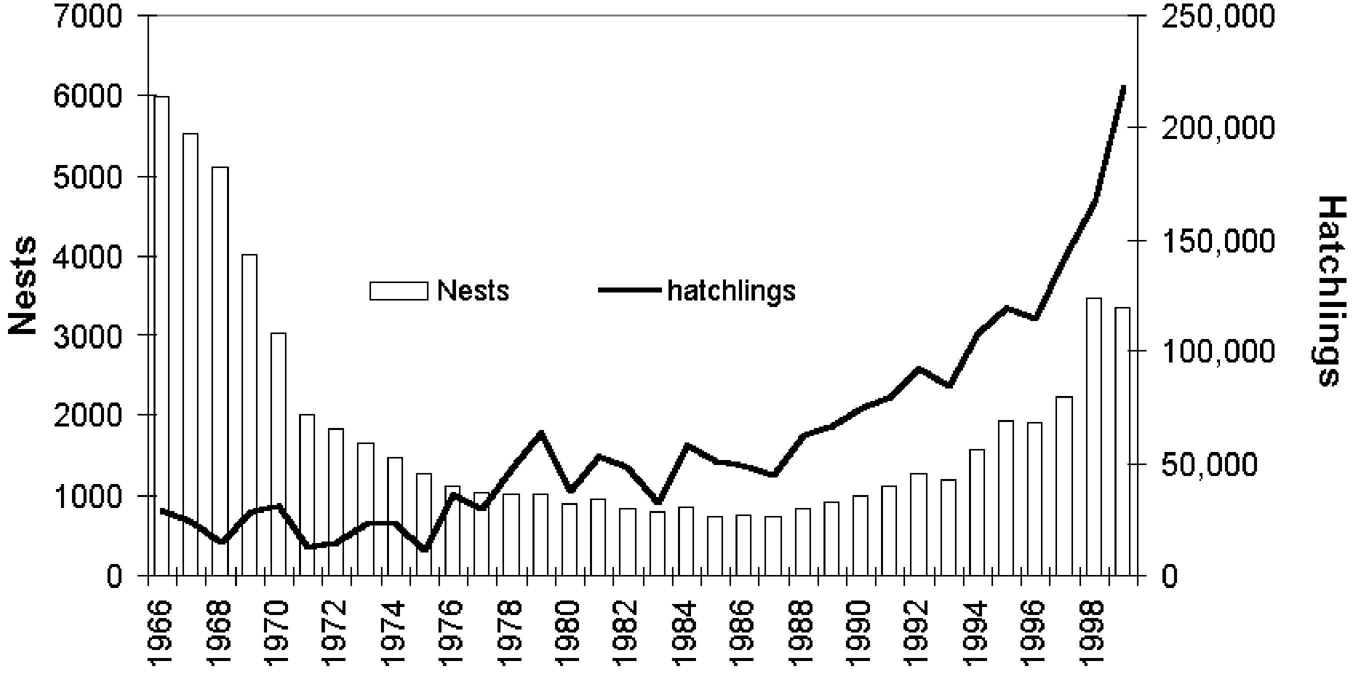


Figure 3. Nests counted at Rancho Nuevo, North Camp and South Camp 1989-1998.

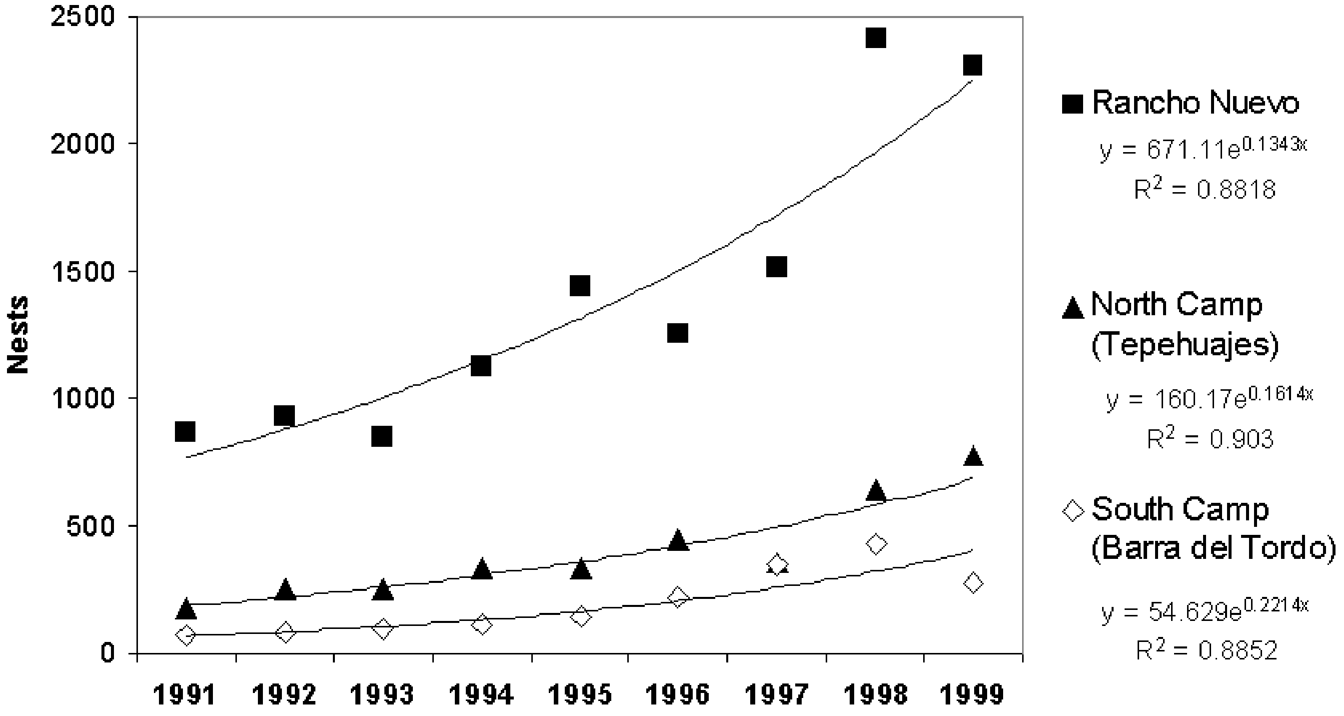


Figure 4. Kemp's ridley nest projection if current trends continue.

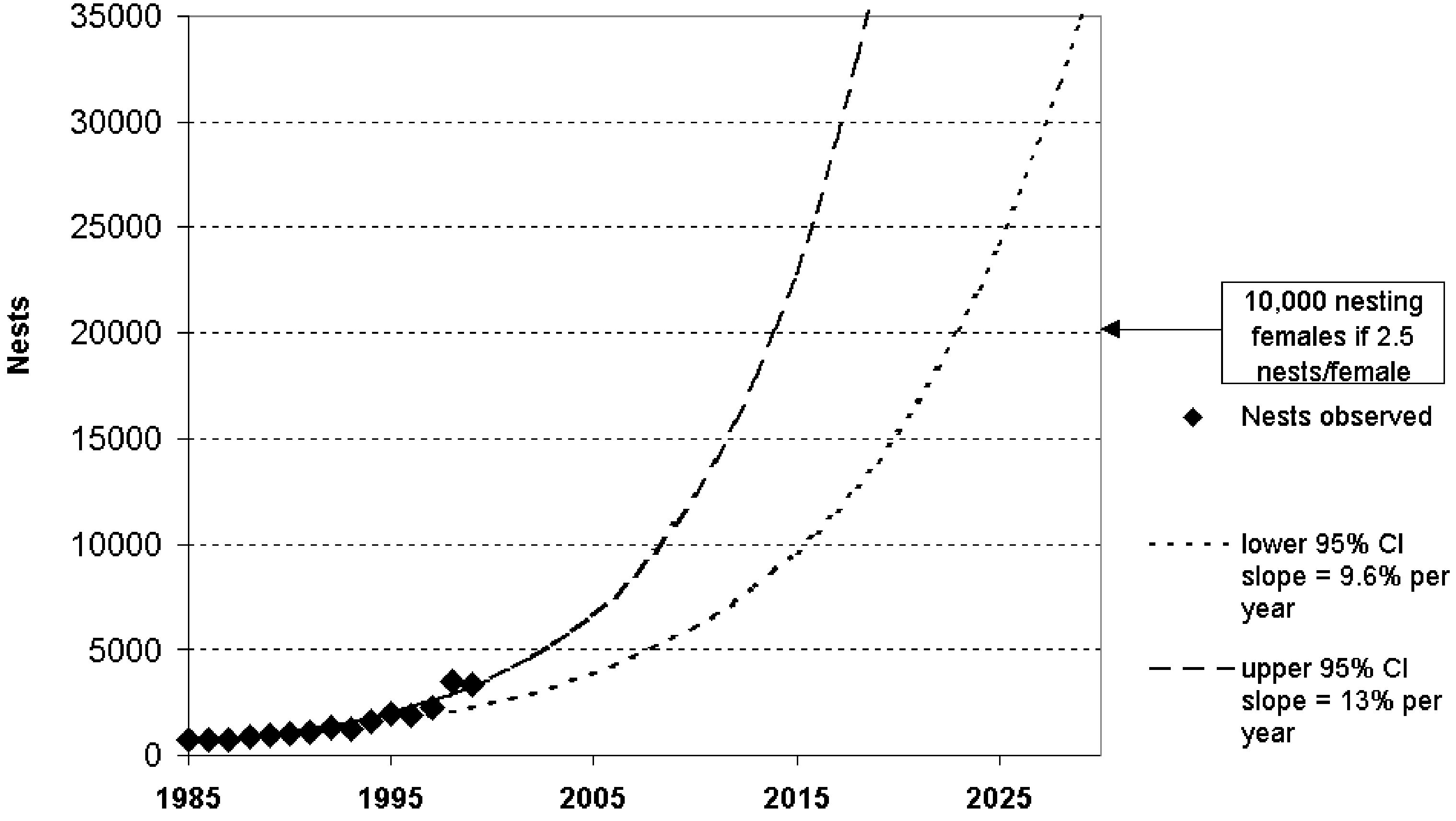
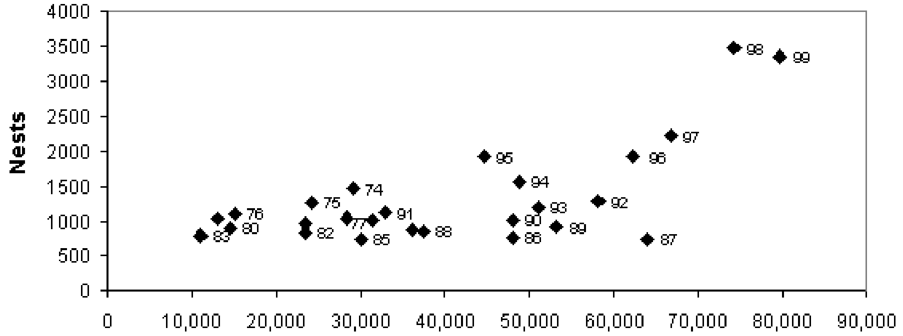
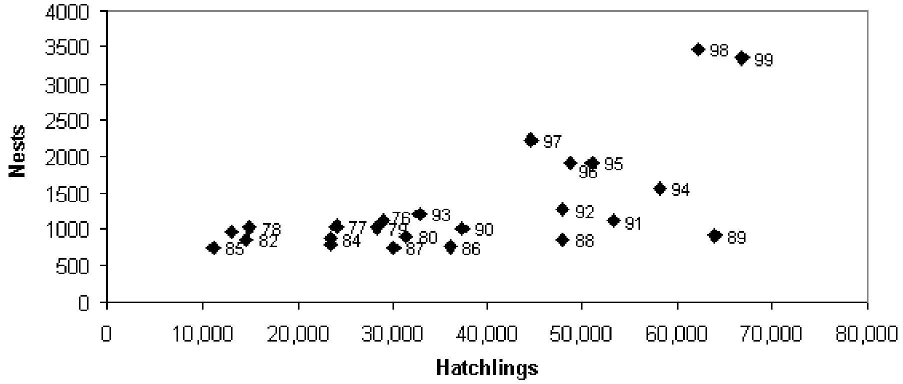


Figure 5. Scatter plots of nests counted vs. hatchlings released 8, 10 or 12 years later. Numbers denote nesting year.

A. 8 year lag



B. 10 year lag



C. 12 year lag

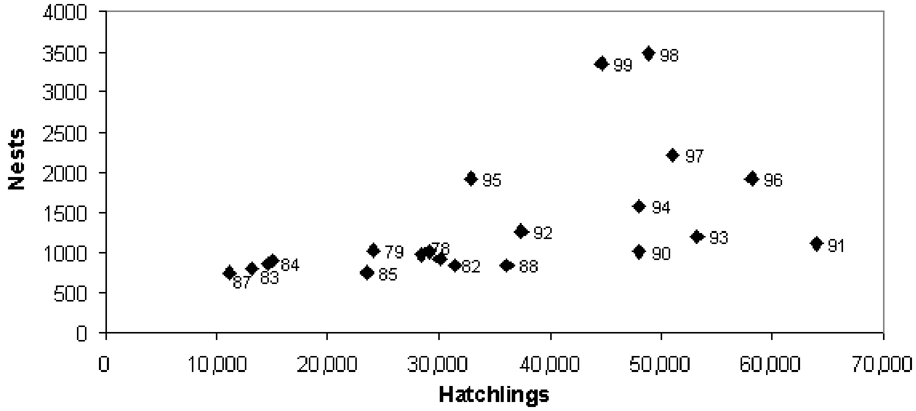


Figure 6. Size frequency distributions of stranded benthic immature Kemp's ridleys in 1986-1987 and 1996-1997, pooled across all statistical zones (Appendix 2).

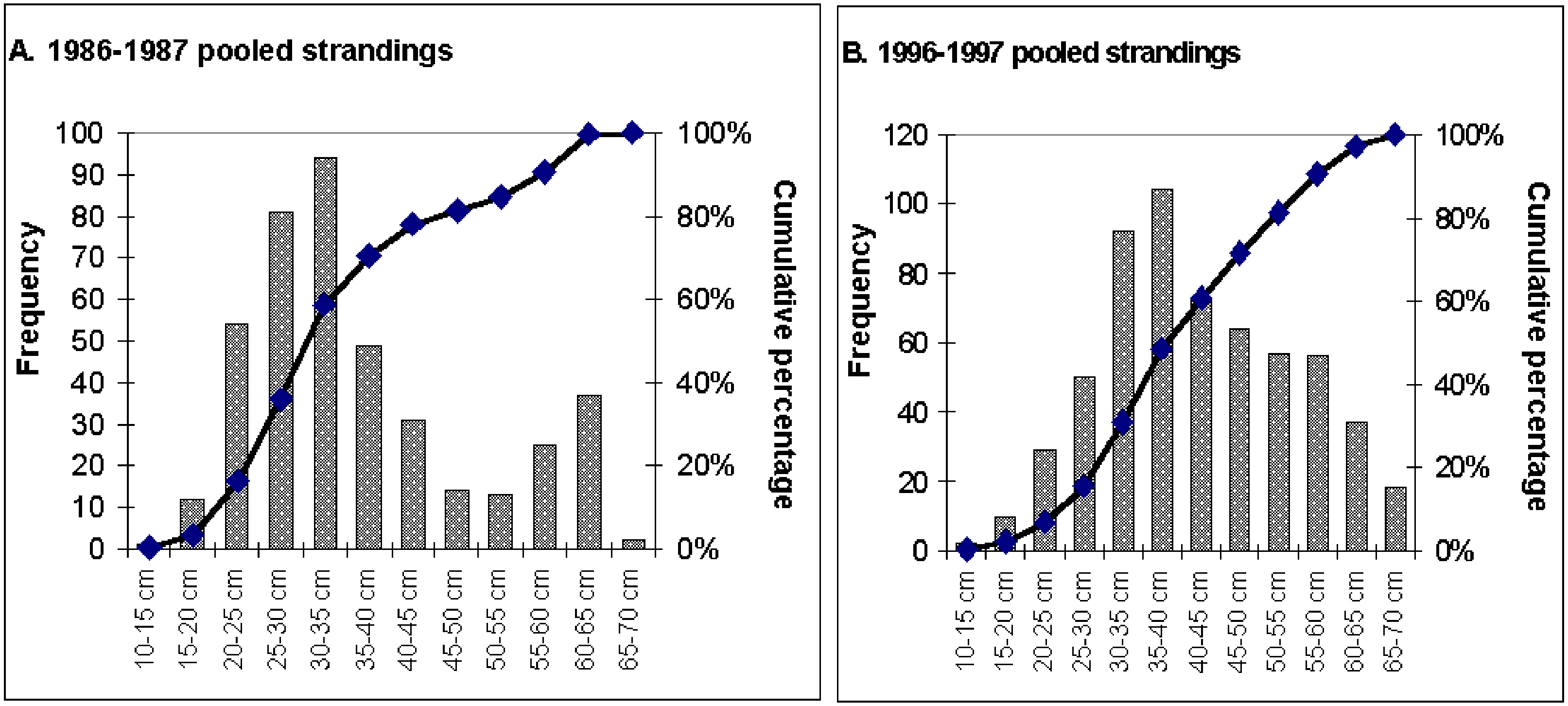


Figure 7. Best fit model estimates of nests for 10 years to maturity, $Z_b = 0.5$, with and without the post-1990 multiplier as a fitted parameter.

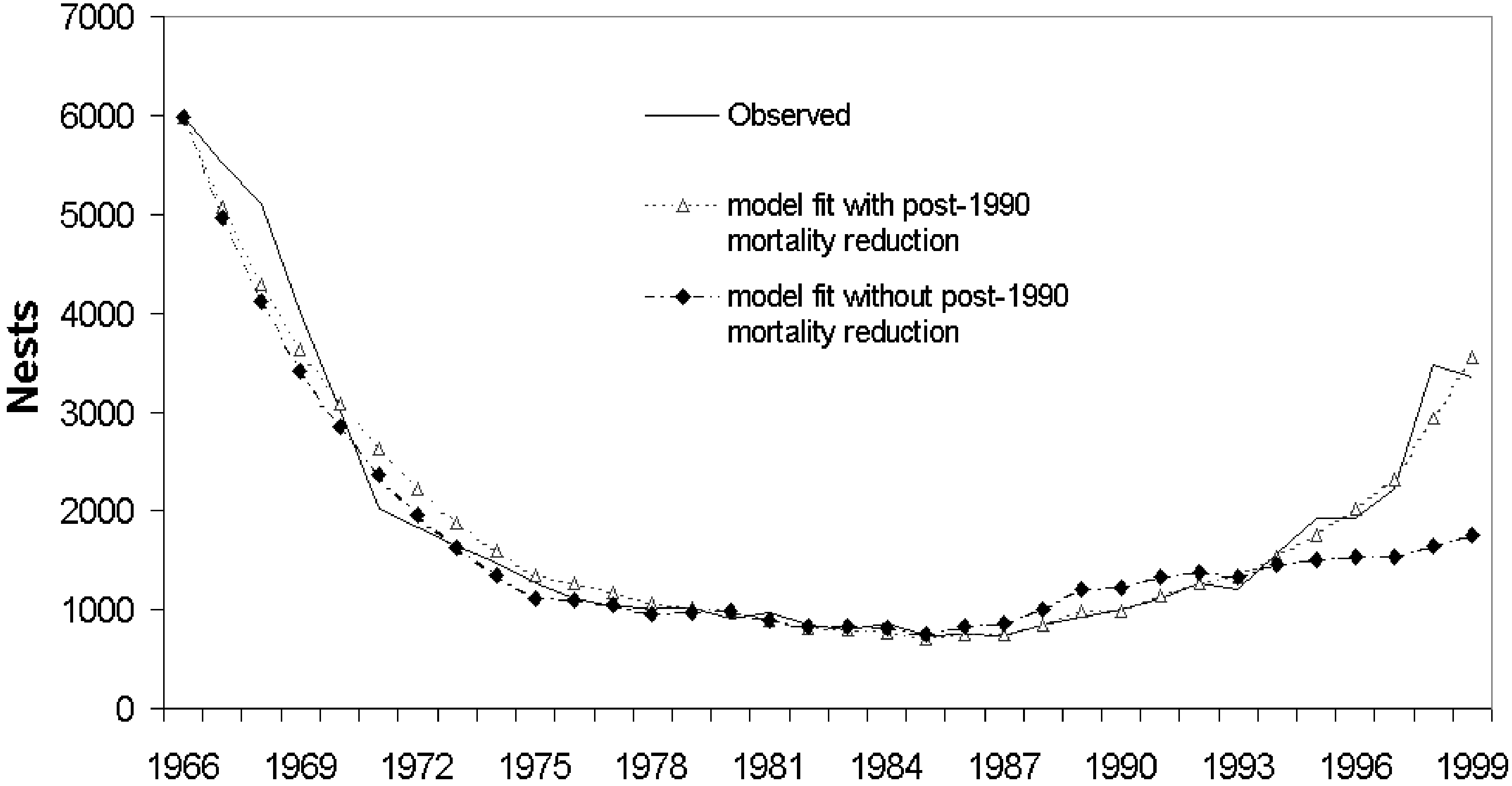


Figure 8. Model fits for 8, 10 or 12 years to maturity with post-1990 multiplier included.

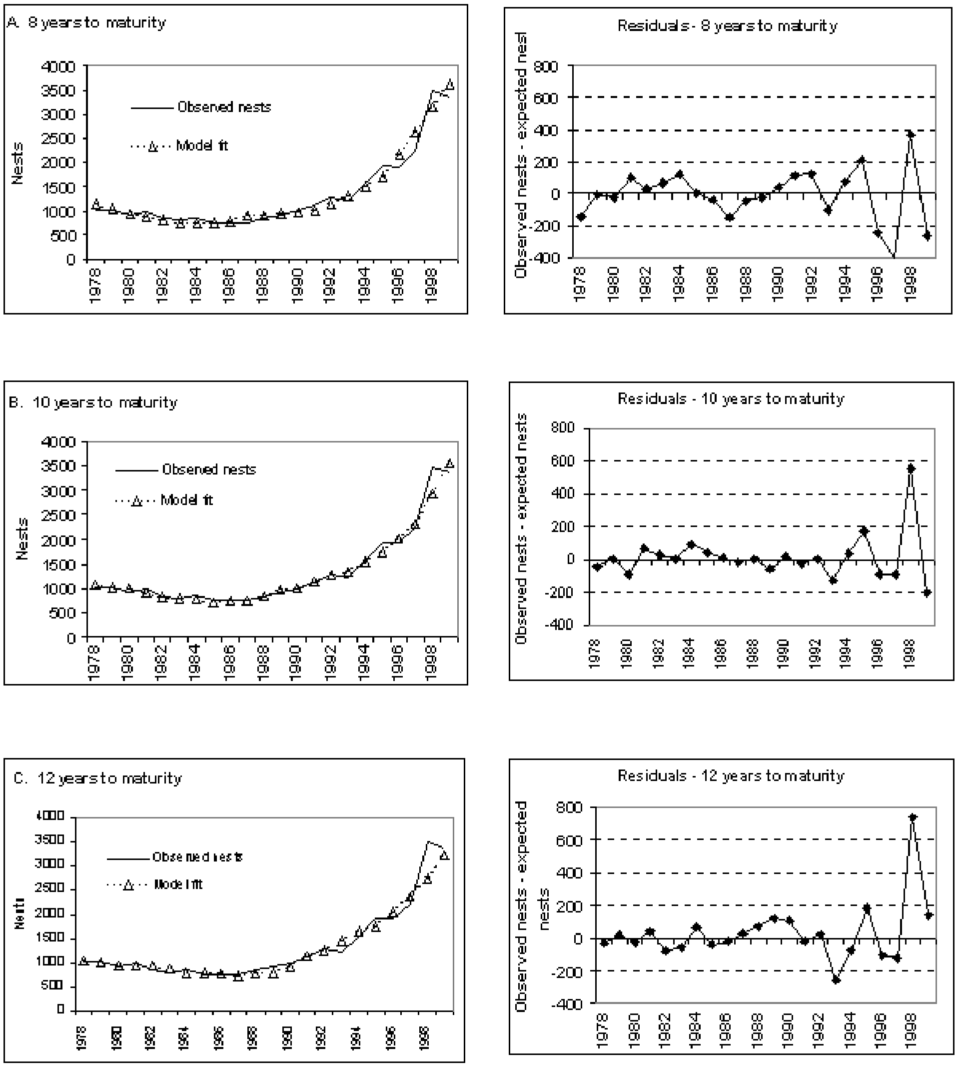


Figure 9. Changes in best-fit parameter estimates for pelagic immature and large turtle instantaneous mortality with changes in the small benthic immature mortality rate, estimated from a catch curve of strandings.

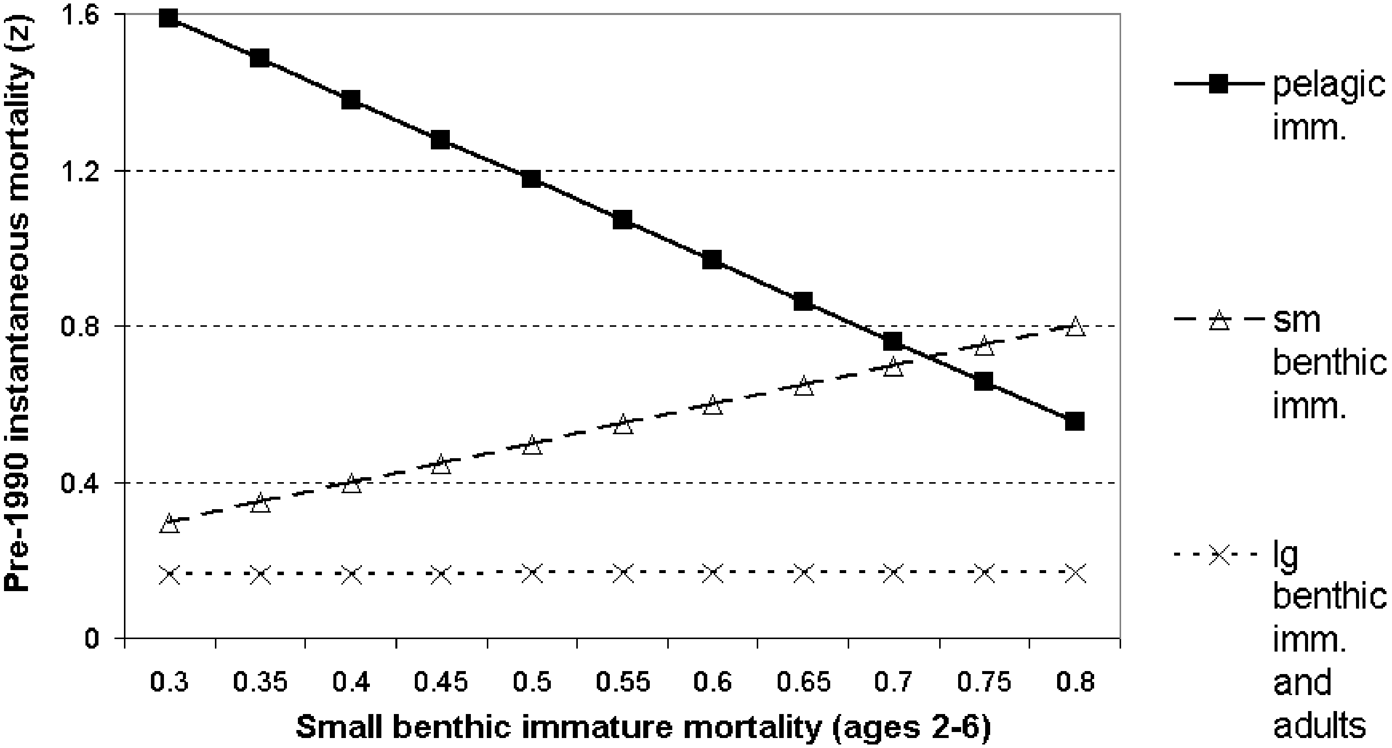


Figure 10. Effect of small benthic immature mortality estimate on the post-1990 multiplier.

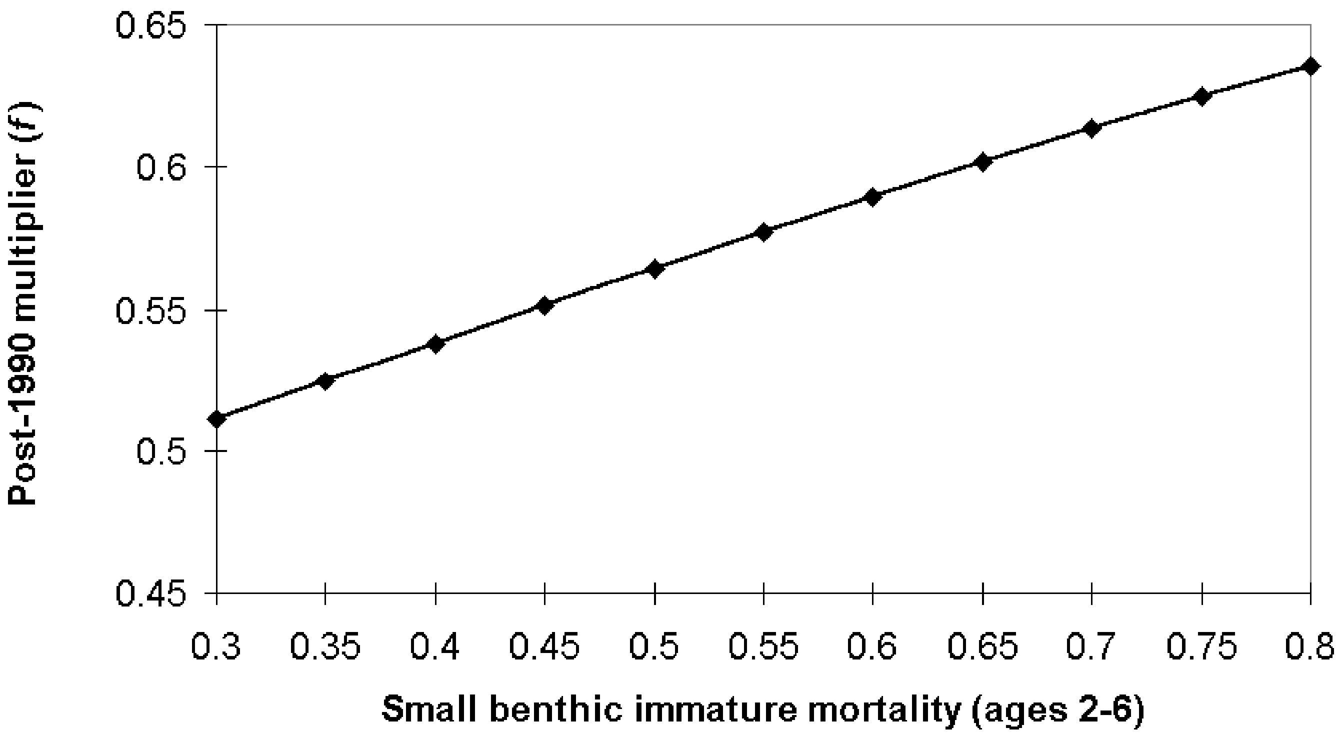


Figure 11. Model estimated population size in 1998, 10 years to maturity model with a range of small benthic immature survival rates. Small benthic immatures should be read with the right axis.

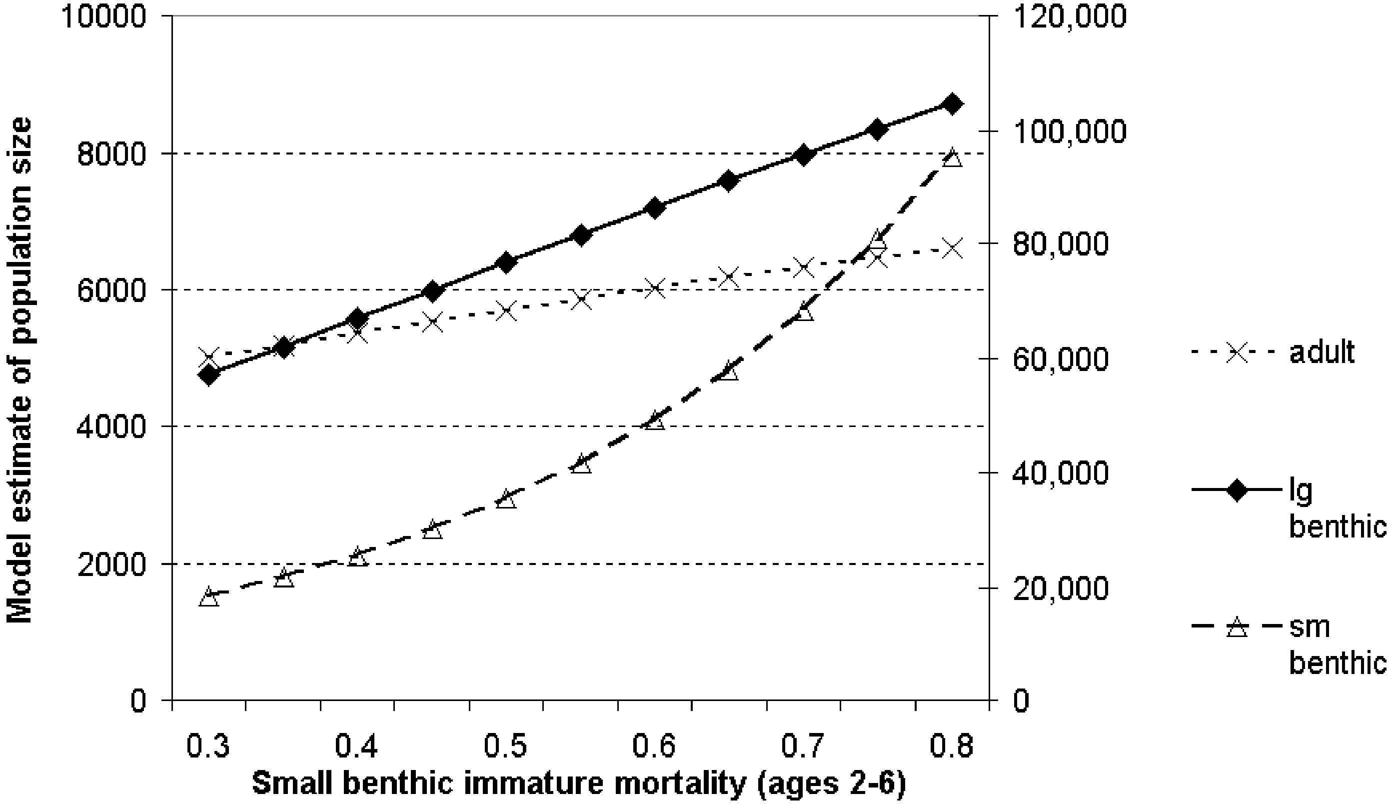


Figure 12. Model fits and residuals for pre-1990 small benthic immature $Z_b = 0.53$, post-1990 $Z_b = 0.34$. A. Model with best-fit post-1990 multiplier for large benthic immature and adult turtles $f = 0.52$. B. Post-1990 multiplier f fixed at 0.65 for all benthic immatures and adults.

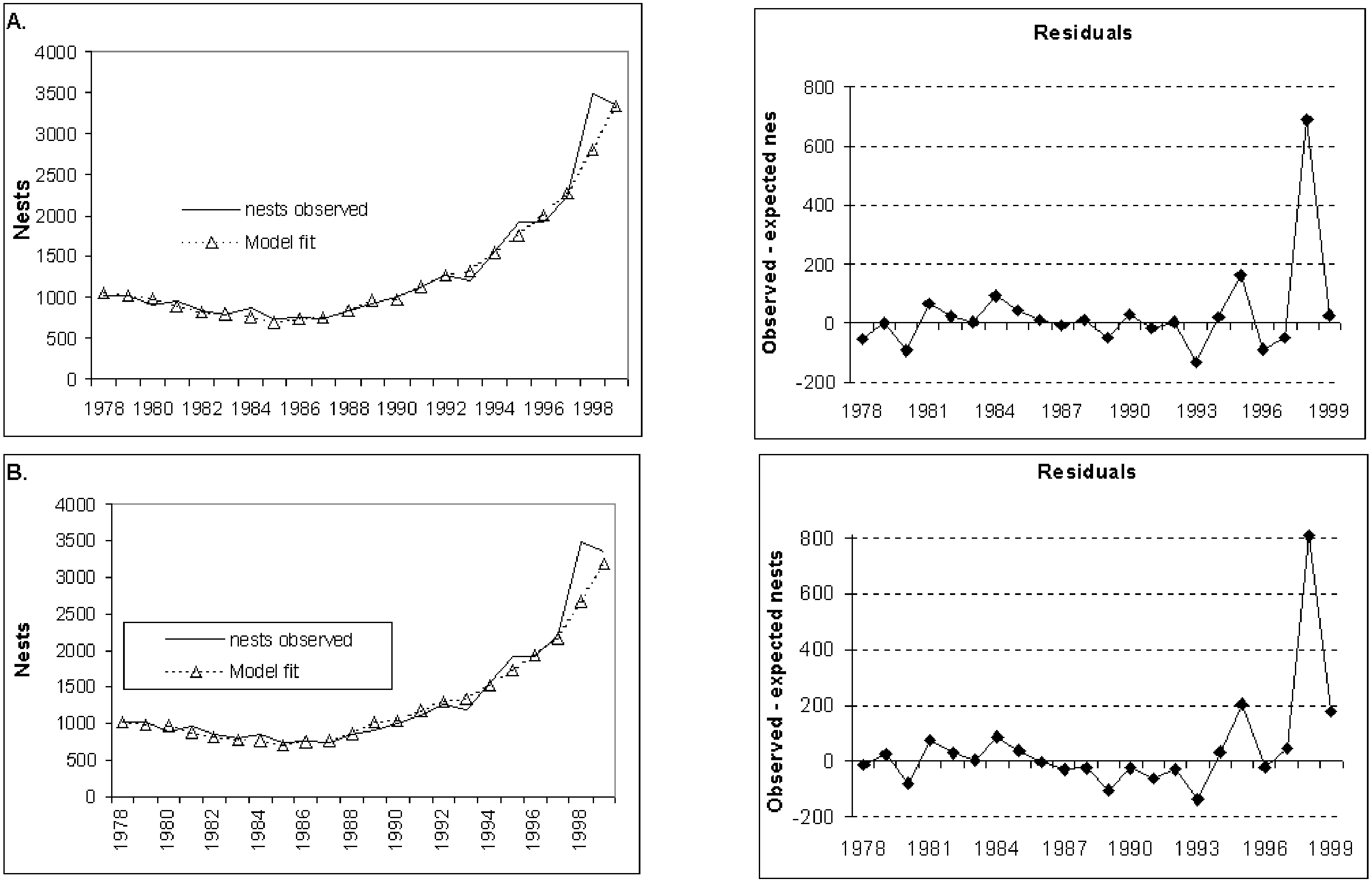


Figure 13. Model projections with $Z_b = 0.5$, assuming mortality rates remain constant after 1990.

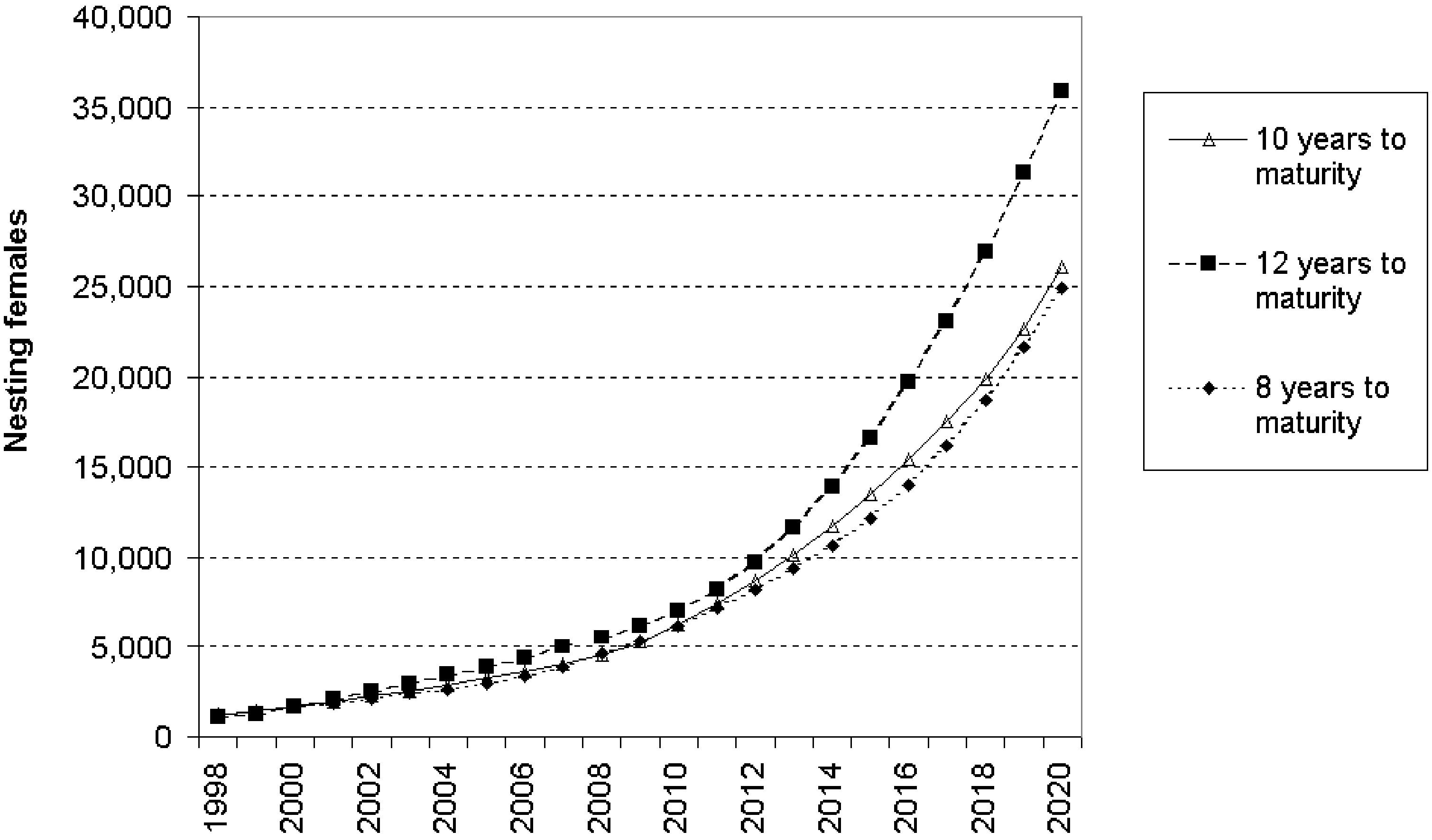
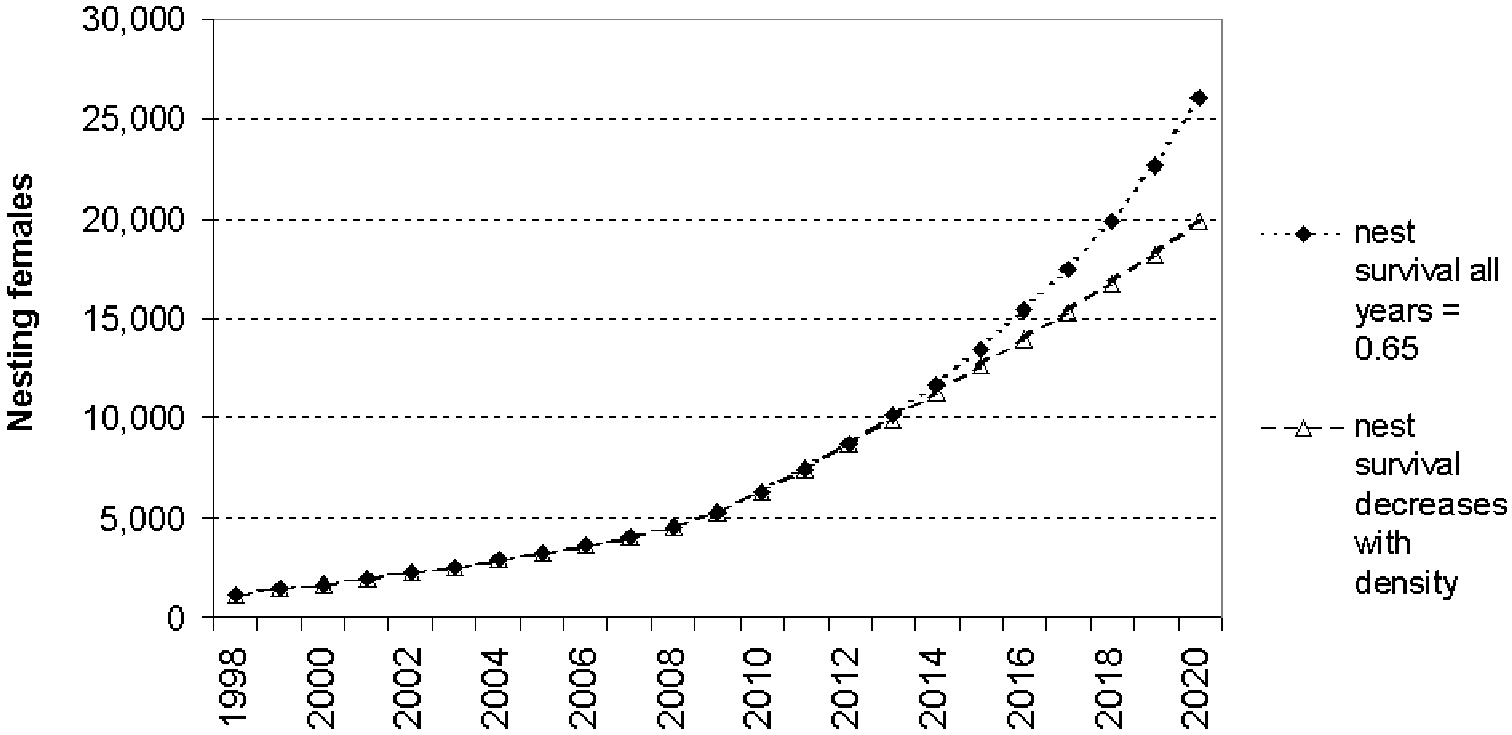


Figure 14. Model projection of nesting female numbers with and without a reduction in nest survival with nest density (10 years to maturity, $Z_b = 0.5$).



Part 2. Loggerhead Status Report

Summary

The loggerhead sea turtle was listed under the Endangered Species Act as threatened in 1978. At the time of listing the species in the North Atlantic was considered to be a single population, but genetic evidence acquired since has indicated the existence of distinct subpopulations which form the basis of management units. The trends of each subpopulation must be assessed separately and the subpopulations may need to be listed separately under the ESA to afford the smaller or declining subpopulations more protection. The TEWG recommends that the life history stages of each loggerhead subpopulation be monitored, and acknowledges that the mix of subpopulations on the foraging grounds makes population-specific assessments complicated. In-water surveys must be conducted to provide data to evaluate the changes in the age structure of the subpopulations.

Stock Definition and Geographic Range

The loggerhead sea turtle (*Caretta caretta*) 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. In the western North Atlantic there are at least 4 genetically distinct loggerhead nesting subpopulations, based on mitochondrial DNA (Bowen *et al.*, 1993; Bowen, 1995; Encalada *et al.,* 1998) (Figure 15) and possibly there are more (Francisco *et al.*, 1999):

1. The Northern Nesting Subpopulation, occurring from North Carolina to northeast Florida, about 29° N. (approx. 7500 nests in 1998).

2. The South Florida Nesting Subpopulation, occurring from 29° N on the east coast to Sarasota on the west coast (approx. 83,400 nests in 1998). 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, FL (approx. 1200 nests in 1998).

4. TheYucatán Nesting Subpopulation, occurring on the eastern Yucatán Peninsula, Mexico (Márquez-M, 1990) (approx. 1000 nests in 1998).

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 more than 35-40% of the global nesting activity of this species (Ross, 1982).

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. 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.

Like other sea turtles, loggerhead post-hatchlings occur in pelagic habitats for a number of years, eventually settling into coastal habitats as benthic immatures, at approximately 40-60 cm SCL in the U.S. (Carr, *1987*). Genetic analyses indicate that turtles from several subpopulations mix on foraging grounds. Table 10 summarizes the estimated proportional contributions of the four identified subpopulations to foraging-ground assemblages, as well as the proportional contributions of Mediterranean haplotypes. The proportional contributions from Chesapeake Bay, South Carolina, Georgia (in part), and the Mediterranean were determined prior to the completion of a more thorough survey of Atlantic nesting beaches and may therefore change by 10% or more, but the qualitative conclusion of mixtures on these foraging grounds remains valid (B. Bowen, *personal communication*). Because South Florida is the largest Nesting Subpopulation contributing to these foraging grounds, the samples are often dominated by turtles of this haplotype. However, the results of these analyses, conducted at multiple foraging sites along the Atlantic seaboard, indicate that mixing of subpopulations is not random and that the contributions of the larger South Florida Subpopulation increase along a general north-south gradient (northeast U.S. to Florida Bay). Analyses of the genetic identities of loggerhead turtles inhabiting foraging grounds of the Gulf of Mexico are underway but not yet available. Preliminary results from analysis of loggerheads stranding in Texas indicate the Northern Subpopulation may account for up to 10% of the animals stranding there, but most come from the S. Florida Subpopulation, along with some from the Yucatán, Mexico Subpopulation (A. Bass, *personal communication*).

Life stage definitions

We have defined the life history stages for loggerhead turtles as: pelagic immature from the time a hatchling leaves the nesting beach until it appears on coastal benthic feeding grounds at around 40-60 cm straight carapace length (SCL); benthic immature from settlement to 92 cm SCL; and adult which is 92 cm SCL[1]. Pelagic immature turtles originating from Atlantic nesting beaches have been observed in the eastern Atlantic near the Azores and Madeira (Maigret, 1983; Carr, 1986, 1987; Bjorndal *et al.,* 1994; Bolten *et al.,* 1994) and in the Mediterranean Sea (Laurent *et al.*, 1998). A growth curve derived from a length-frequency analysis of pelagic immatures in the Azores indicates that loggerheads may stay in the North Atlantic Gyre for 7-12 years, depending on the size of recruitment to the benthic stage (Bjorndal *et al.*, 2000*)*. It has been suggested that turtles that originate from Gulf of Mexico beaches also utilize the North Atlantic Gyre and are distributed to it via the Loop Current of the Gulf of Mexico and Florida Current (Collard and Ogren, 1990). Strandings records indicate that when pelagic immature loggerheads reach 40-60 cm SCL they recruit to coastal inshore and nearshore waters of the continental shelf throughout U.S. Atlantic and Gulf of Mexico. Benthic immatures have been found from Cape Cod, Massachusetts, to southern Texas, and occasionally strand on beaches in northeastern Mexico (R. Márquez-M., *personal communication*). Large benthic immature loggerheads (70-91 cm) represent a larger proportion of the strandings and in-water captures (Schroeder *et al.*, 1998) along the south and western coasts of Florida as compared with rest of the coast, but it is not known whether the larger animals actually are more abundant in these areas or just more abundant within the area relative to the smaller turtles. Benthic immature loggerheads foraging in northeastern U.S. waters are known to migrate southward in the fall as water temperatures cool (Epperly *et al.*, 1995; Keinath, 1993; Morreale and Standora, 1999; Shoop and Kenney, 1992), and likely migrate northward in spring. Given an estimated age at maturity of 21-35 years (Frazer and Ehrhart, 1985; Frazer *et al.*, 1994), the benthic immature stage must be at least 10-20 years long. The adult life history stage, 92 cm straight carapace length, has been reported throughout its U.S. range and throughout the Caribbean Sea. Nesting occurs primarily from North Carolina southward to Florida with assemblages recorded in the Florida Panhandle and on the Yucatán Peninsula (Figure 15). Non-nesting females are reported from the entire area and little is known about the distribution of adult males which are seasonally abundant near nesting beaches during the nesting season.

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[1] Appendix 4 of our first report (TEWG, 1998) details how we arrived at the decision to use 92 cm SCL to distinguish benthic immature and adult turtles. It was based on an arithmetic mean which may be an inappropriate measure of central tendency in non-normally distributed data, which some nesting beach data appear to be. Also, we have not considered the possibility that the average size of neophyte nesters may be subpopulation-specific.

Status and Trends

Nesting beaches

Northern Subpopulation

The addition of nesting data from the years 1996, 1997, and 1998, did not change the assessment of TEWG that the number of loggerhead nests in the Northern Subpopulation is stable or declining (Figures 16 and 17). The annual number of recorded nests for the period 1989-1998 ranged 4370-7887 nests/year (Table 11). No trends are detectable for North Carolina, South Carolina or Georgia during that period; nests in northeast Florida may be increasing, although the data are too variable to detect a significant trend (Table 12). A longer time series of nesting data for a portion of the Northern Subpopulation comes from Cape Island at the Cape Romain National Wildlife Refuge, South Carolina (Figure 16B). There, the number of recorded nests for the period 1975-1998 ranged 579-2654 nests/year. Nesting at Cape Island has declined an average of -2.7% per year over the 24-year period (r-square=0.284, P = 0.007), but nearly all of that decline occurred during the 1970's. Since 1989, the Cape Island nest numbers have varied annually with no significant trend (1989-1998 slope = -1.4% per year, r-square = 0.023, P = 0.675). Annual nesting at Cape Island was found to correlate well with nesting in North Carolina, South Carolina (remaining beaches), Georgia and the Northern Subpopulation as a whole for the period 1989-1998, but did not correlate as well with nesting activity of the South Florida and Florida Panhandle Subpopulations (Table 13). The residuals from the regression model correlated well with the residuals from northern nesting beaches and southeast Florida, but not the SW Florida nor the Panhandle residuals.

South Florida Subpopulation

The addition of nesting data from the years 1996 through 1998 did not change the assessment of TEWG that the number of loggerhead nests in the South Florida Subpopulation is stable or increasing. The annual number of recorded nests for the period 1989-1998 ranged 48,531-83,442 nests/year (Table 11), with an average rate of increase over the time series of 3.6% per year (r-square = 0.461, P = 0.031, Table 12). The time series suggests a 4-year cycle in nesting, although the time series is too short to confirm this (Figure 17A). A longer time series of nesting data for a portion of the South Florida Subpopulation comes from Hutchinson Island, Florida (Figure 17B). There, the number of recorded nests for the period 1981-1998 ranged 3121-8214 nests/year and had a mean rate of increase of 4% per year (r-square = 0.746, P < 0.001). In more recent years (1989-1998), Hutchinson Island nests have increased 2.7% per year but the trend is not significant because of data variability (r-square = 0.308, P = 0.096). Variation in nest counts at Hutchinson Island seem to support the 4-year cycle hypothesis for recent years but not earlier years in the time series (Figure 17B). The nest count was found to correlate extremely well with nesting in the remaining parts of the southeast Florida region for the period 1989-1998, but did not correlate as well with the nests counted at northern beaches nor those in the Florida Panhandle (Table 13). However, the residuals from the ln-transformed regression correlated fairly well with the residuals from all areas except the Panhandle.

Florida Panhandle Subpopulation

The number of loggerhead nests recorded for the Florida Panhandle Subpopulation is increasing, but much of this increase is due to expanded beach monitoring. The length of beach monitored in the Panhandle increased from 186.5 km in 1989 to 359.9 km in 1997. Thus, the status of this Subpopulation cannot be determined at this time.

Nest count correlations across states and regions

Table 14 gives the Pearson correlation coefficients for loggerhead nest counts in all U.S. areas. A second set of correlations was calculated with ln-transformed data residuals to remove the linear trend of each time series. State nest counts within a nesting subpopulation generally correlate well with each other and the subpopulation totals. However, the three nesting subpopulations are less well correlated with each other over the 10 year time period. Nests in North Carolina and Georgia may follow the 4-year pattern observed in southeast Florida (Figures 16 and 17), but South Carolina nests are much more variable. The Florida Panhandle Subpopulation is poorly correlated with the other two subpopulations, likely because much of the recent increase in nest counts for that area are due to an increase in monitoring effort. However, the Panhandle nests and SW Florida nests are highly correlated.

Yucatán nesting subpopulation

There are few nesting surveys for loggerheads in Mexico. There are 8-12 turtle camps operated each year in central Quintana Roo on the Yucatán Peninsula. In 1998, 1052 nests were recorded at 12 camps on Quintana Roo beaches (Xcaret, 1999). Nesting of this Subpopulation is believed to extend into Belize and nesting appears to be stable or increasing (R. Márquez-M., *personal communication*).

Loggerhead nesting elsewhere in the Gulf of Mexico

Nesting of loggerheads throughout the Gulf of Mexico is sporadic outside of Florida and few areas conduct systematic surveys of nesting beaches (S. MacPherson, *personal communication*). The first estimate of nesting in the Gulf of Mexico outside of Florida indicated that between 1960 and 1962 a reproductive assemblage of approximately 100 loggerheads nested on the remote beaches of the offshore islands of Louisiana, Mississippi, and Alabama (Ogren, 1978). Aerial surveys in 1977 of the U.S. Gulf beaches west of Florida indicated that nesting had declined (*Ibid*). In recent years about 21-31 nests were found annually at Bon Secour National Wildlife Refuge in Alabama (1994-1998) and a couple are found each year on Dauphin Island (1995-1997) (Mays and Shaver, 1998; S. MacPherson, *personal communication*). Up to 9 nests were found annually at the Gulf Islands National Seashore in Mississippi 1990-1998 and less than 10 nests were found annually at the Breton National Wildlife Refuge in Louisiana (1989-1990) (*Ibid*). There are few records of loggerheads nesting on the Texas Coast: 18 since 1977 (D. Shaver, *personal communication*). Eleven of the 18 recorded nests occurred in 1996-1998 and all but one was on the southern Texas coast. Loggerheads also nest occasionally in the

Mexican states of Tamaulipas, Campeche, and Yucatán. During this decade the annual nest counts for these areas were: 1-5 nests at Rancho Nuevo and Ostionales in Tamaulipas, about 50 nests in Campeche, and about 100 nests in Yucatán (R. Márquez-M., *personal communication*). The genetic identity of these scattered nests is unknown.

Strandings

Complete strandings information by size class, statistical zone and year are provided in Appendix 2. As with the analysis of strandings of Kemp's ridley turtles, the loggerhead strandings used by the TEWG excluded incidental captures, post-hatchlings, cold-stunned or head-started animals. Figure A2.1 in Appendix 2 shows the statistical zones for which sea turtle strandings are reported. Monitoring effort is not directly comparable between zones but has been reasonably consistent over this period. Figure A2.3 a, b, and c depicts the loggerhead strandings reported by statistical zone, by year, 1986-1997. There is no survey effort in zones 15 and 16, due to inaccessibility of shoreline, and coverage is low in zones 13 and 14. In the eastern Gulf of Mexico (zones 1-12, partial 24-25), survey coverage is low in zones 1, 3, 6, and 7 due to inaccessibility and zone 2 has very little land mass. The lack of data from these zones may or may not reflect a lack of strandings. Along the southeast U.S. Atlantic coast, coverage is also low in zones 24 and 25. In the northeastern U.S. Atlantic, survey coverage is less rigorous. However, high human densities along the coast in this area suggests most strandings will get reported. This is not true for inshore waters, such as the Chesapeake Bay and Pamlico and Core Sounds of North Carolina, where strandings likely go unreported.

Trends

Table 15 shows the loggerhead strandings by region. Over the 12-year period 1986-97, the southeast accounted for 63% of the strandings totals, with the Gulf of Mexico and northeast Atlantic accounting for 21% and 15%, respectively. Strandings in the southeastern U.S. and the Gulf of Mexico declined in the early 90's, particularly 91-93, but have increased since then. The combined total for 1995-97 was 13% higher than the combined total for 1986-88 (Table 16). In particular, strandings in the northeast more than doubled over that time frame, increasing the relative importance of mortality in the northeast (from 10 to 19%). The northeast is the only region which has shown significant trends throughout the 1986-1997 period. However, since 1991 all three areas have shown significant increasing trends in strandings, 10-14% per year. Overall, loggerhead strandings showed an average annual increase of 10.5% per year for the period 1991-1997.

Strandings "hot spots"

While loggerhead strandings occur throughout the eastern U.S., the largest proportion of them occur along the southeast U.S. Atlantic coast (Figure A2.3, Appendix 2). Strandings are high throughout the area, with Zone 30 highest in most years and Zone 26 and the southern segment of Zone 36 lower in most years. Adults strand throughout the region. Strandings are lower in the Gulf of Mexico. Zones 4 and 5 have somewhat higher stranding numbers; of particular concern is the proportion of adult loggerheads stranding in these zones. Zones 18, 19 and 20 also have higher strandings but with few adults. In the northeast U.S. Atlantic loggerhead strandings are concentrated in Zones 36 and 37 with fewer adults.

Based on NMFS estimates shrimping intensity is much higher in the western Gulf of Mexico than in the east. It is concentrated nearshore in the west; more than half the shrimping effort is the 0-18m depth zone. In the eastern Gulf, effort peaks in the 18-37m zone (>60% of the effort occurs at these depths) (Nance, 1992, 1993; McDaniel *et al.*, 2000). Loggerhead strandings are consistently highest in zones 4-5 (West Florida) and 18-20 (Texas). Strandings in West Florida include a high proportion of adults whereas strandings in Texas are exclusively juveniles. Given the relatively low shrimping effort in the eastern Gulf, the impact on loggerheads (particularly adults) appears high.

Systematic surveys for sea turtle strandings do not occur in Mexico but a few (<10) loggerhead strandings have been recorded at Rancho Nuevo in recent years. These have been primarily large benthic immatures (>63 cm CCL), except for one 95 cm CCL animal in 1997.

TED effect analyses

Biweekly strandings data from South Carolina and Georgia (1980-97) were analyzed to determine the significance and magnitude of TED effects in reducing strandings (Royle and Crowder, 1998). The methods used were similar to those applied to the S.C. data (1980-93) in which strandings were shown to be reduced by 44% (Crowder *et al.*, 1995). TEDS significantly reduced strandings over the period 1980-97 by an estimated 40% in South Carolina and 58% in Georgia relative to strandings estimates without TEDs (Royle and Crowder, 1998). A significant “TED effect” on strandings is detectable through the time series analysis of biweekly data in spite of the increasing trend in annual strandings. Recent analysis of the Georgia data utilizing Poisson regression and taking into account shrimp landings indicates strandings in Georgia (per unit shrimp catch) were reduced 37% when TEDs where in use compared to periods when TEDs were not in use (Royle, 2000).

In-water surveys

Fishery independent, in-water studies of sea turtles have been carried out at multiple sites in the Gulf of Mexico and U.S. Atlantic. The primary goals and objectives of these studies vary, and the species targeted vary as well. Some studies have been conducted for many years (10+), others have only recently been initiated and still others have been intermittent or aperiodic. There are numerous difficulties in designing in-water studies to ensure that standardized catch per unit effort (CPUE) methods are employed such that data can be compared, not only within a site but also site-to-site. Environmental conditions including depth strata, current patterns, visibility, water temperature, and investigator experience can greatly affect catch rates. Biological conditions, such as prey abundance, complicate sampling strategies that may be based on random distribution of the target species. Changes in effort, variability in temporal sampling, and different sampling methodologies make analyses of population trends particularly difficult. At this time, only two studies enable relatively long-term comparisons of CPUE on a within study site basis[2]. These two sites are the nearshore waters off the south central east coast Florida at the St. Lucie Nuclear Power Plant (Quantum Resources, 1995, unpublished data) and the Indian River Lagoon along the central east coast of Florida at Sebastian Inlet (Ehrhart *et al.*, 1996). The TEWG does not feel that changes in CPUE evaluated with recent data from the naked net survey conducted by the Gulf & South Atlantic Fisheries Development Foundation, Inc. (1998; Jamir, 1999) indicate a real 8-fold increase in population size in the southeast U.S. because of the differences in sampling as discussed in the Kemp's report above, but that it does indicate there has been an increase of an unknown magnitude.

Ehrhart *et al.* (1996) analyzed loggerhead CPUE data from his central Indian River Lagoon study site for three combined sampling periods (1983-1985, 1988-1990, and 1993-1995). The turtles were captured by gill nets strung across the lagoon, with CPUE measured as turtles caught per kilometer net hour. Ehrhart concluded that although there was some variation between the three spans of years, CPUE was not statistically different (Kruskal-Wallace = 1.244, df = 2, P = 0.5369). Captures of loggerheads at the Indian River Lagoon site are immature turtles and hence the CPUE statistics would not be affected by influxes of adult turtles during the nesting season.

The intake pipe of the St. Lucie Nuclear Power Plant captures sea turtles each year, depositing them relatively unharmed in an enclosed lagoon. The species and size of each turtle is recorded, after which the turtles are tagged and released. Because the intake pipe operates continuously, the capture effort may be considered relatively constant. Turtles of all size classes have been caught, but it is not known whether the St. Lucie turtles represent an unbiased sample of the nearby turtle population. Over the entire time series (1977-1998), the exponential trend in total loggerheads captured at St. Lucie has a slope of 4.2% per year (r-square = 0.40, P = 0.002) (Table 17). However, the early part of the time series shows little overall growth and a potentially cyclical pattern in abundance (Figure 18). In recent years, the trend has increased to 17% per year (r-square = 0.76, P = 0.005), although the number of loggerheads captured in 1997 dropped dramatically. The TEWG (1998) reported significant increases in numbers for adult loggerheads (92 cm SCL) at St. Lucie which track increases in the nesting population, as determined by nesting surveys. These trends continue when post-1995 data (through 1998) are included (Figure 18, Table 17). Increased numbers of benthic immature turtles captured in 1996 and 1998 resulted in a significant increasing trend for these turtles as well, which was not detected in the 1977-1995 time series due to data variance (TEWG, 1998). Adult and immature captures were generally well correlated across years (Pearson's r 0.65, P<0.05), without evidence of a time lag.

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[2] Recent examination of SEAMAP trawl data collected by South Carolina Department of Natural Resources indicates that those data may also be a source of trend information. The extensive trawl survey is based on a stratified random sampling design in nearshore waters between Cape Hatteras and central Florida and has been conducted seasonally since 1989.

The two studies which provide the longest and most consistent time series of data do not agree in their assessments of trends in the population of benthic immature loggerheads. Because juvenile abundance and capture probability are highly variable, it will be difficult to detect trends without long time series from multiple sites. Both of the surveys discussed here consist of samples that are likely to be dominated by the South Florida Subpopulation, thereby giving little indication of trends in benthic immature loggerheads originating from the Northern Subpopulation. A survey of benthic immature turtles in Pamlico and Core Sounds, North Carolina, will provide more information regarding trends in the different subpopulations. More comprehensive surveys are needed to monitor the status of benthic stages of the population and multiple, index area studies that yield statistically valid CPUE data are urgently needed. Trawl surveys may be one method of accomplishing this goal. The Working Group recommends the design and implementation of a pilot trawl survey, bearing in mind the patchy nature of population distribution (see Research Recommendations).

Sex ratio

Studies of loggerheads along the Atlantic coast suggest that these populations exhibit a female bias. Laparoscopy of 47 turtles examined during fall 1997 in North Carolina found 68% were female and 32% were males (Braun-McNeill *et al.*, 2000). Serum androgen sexing of turtles captured in waters 20 ° C in the same area, 1996-1998 indicated 72% female, 27% male, and 1% unknown (n=114) (Braun-McNeill et al., *in press*). Wibbels *et al.* (1987) reported a female bias (66%), based on serum testosterone titres, across Atlantic coast sites sampled during the 1980's: Chesapeake Bay (52%), Cape Canaveral (60%), Hutchinson Island (74%), and Indian River (58%). It can be speculated that female-biased sex ratios are likely given the predominance of turtles from the South Florida Nesting Subpopulation, whose nest incubation temperatures are generally higher than the pivotal temperature required for an even sex ratio in loggerheads (Mrosovsky, 1988).

Size distributions

Mean size of stranded loggerheads 1986-87 vs. 1996-97

The mean size of benthic immature loggerheads that stranded from 1986-1997 varied from region to region with the largest turtles in the eastern Gulf (about 80 cm SCL) and somewhat smaller turtles elsewhere (about 60-70 cm SCL). One might expect that a growing population would experience a shift in size distribution towards smaller turtles. We found no evidence of a shift in the mean size of loggerheads stranded over this period, but the size distribution of stranded loggerheads was significantly different in 1996-97 compared with 1986-87 (Kolmogorov-Smirnov Test, P<0.0001) (Figure 19). Benthic juvenile strandings in the 55-65 cm SCL size range increased in abundance in the 1990s relative to the 1980s.

Size distributions from the St. Lucie Power Plant

The size composition of loggerheads captured at the St. Lucie nuclear power plant has varied from year to year with some long-term trends (Figure 20). In the 1980's, the proportion of small immatures captured decreased, becoming as low as 50% in 1989 and 1990. Since 1991, this proportion has stabilized at around 70%. The proportion of adult loggerheads in the capture records reached a peak of 17% in 1990 and has fluctuated since 1991 (5-13.5%). The average proportions for 1991-1998 are 69% small immatures, 22% large immatures and 9% adults.

Population size estimates

Nesting data collected on index nesting beaches 1989-1998 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 an average of 4.1 nests per nesting female, Murphy and Hopkins (1984)) and of the number of adult females in the entire population (based on an average remigration interval of 2.5 years, (Richardson *et al.*, 1978). The total number of nests laid along the U.S. Atlantic and Gulf coasts, 1989-1998, has ranged from 53,016-89,034 annually, representing, on average, an adult female population of 44,780 (Table 11) [(nests/4.1) * 2.5]. On average, 90.7% of the nests were from the South Florida Subpopulation, 8.5% were from the Northern Subpopulation, and 0.8% were from the Florida Panhandle Subpopulation.

The ratio method used by TEWG (1998) to estimate minimum population size of post-pelagic loggerhead turtles in nearshore U.S. waters required assumptions about the accurate representation of loggerhead size classes in the stranding data. A recent, cursory appraisal of capture data and of TED design information leads the TEWG to believe that this assumption may not hold true. Data from the naked-net study conducted by the Gulf & South Atlantic Fisheries Development Foundation, Inc. (1998; Jamir, 1999) show a significantly smaller (Kolomogrov-Smirnov Test, P<0.005) proportion of larger loggerheads (large immatures and adults) than appears in the stranding data (Table 18). A size bias in the stranding data might be expected if large loggerheads are less likely to be excluded by TEDs than small loggerheads and because of this, they incur higher mortality and are more likely to strand. A recent examination of loggerhead morphometrics and TED exit opening dimensions shows that loggerheads greater than approximately 70 cm may not be readily excluded by TEDs (Epperly and Teas, 1999). Also, nesting female loggerheads that occur in nearshore habitats may be more prone to incidental capture and repeat captures, thereby increasing their mortality rate and probability of stranding. In light of this information, we have chosen not to recalculate population size estimates using the ratio method.

Conclusions from nest numbers, strandings, and in-water surveys

In a species with a long age to maturity, such as the loggerhead, nesting trends alone may give an incomplete picture of population status. It is conceivable for a population with no new recruitment to the benthic juvenile stage to continue to show increases in nesting for a number of years as benthic juveniles from past cohorts mature. Conversely, a population could continue to show declines in nesting over time due to losses of adults while the immature population is increasing. Thus multiple lines of evidence must be considered in order to determine true population status.

The loggerhead recovery plan established recovery goals primarily in terms of nesting activity. While the loggerhead recovery plan was drafted before the mitochondrial DNA evidence of distinct nesting subpopulations became available, goals in terms of numbers of nests per season were given for North Carolina, South Carolina and Georgia, roughly encompassing the range of the Northern Subpopulation, with a goal for Florida of an increasing adult female population.

Taken together, the data reviewed by the TEWG would suggest that the South Florida Subpopulation of loggerheads is increasing, while no trends are evident in the Northern Subpopulation in recent years. Long term trends at Hutchinson Island, an index nesting beach for the South Florida Subpopulation, has increased by approximately 4% per year since 1981. While stranding data and trends cannot be attributed to subpopulations, the size of the South Florida Nesting Subpopulation taken together with the genetic evidence that animals from this Subpopulation disperse along the entire Atlantic coast suggest that it predominates in the strandings. Strandings have increased in recent years and there is evidence of a shift in size class distribution toward smaller turtles, supportive of an increasing population. Again, the in-water data are not conclusive, but apparent increases at the St. Lucie Power Plant, where in 1999 an estimated 69% were from the South Florida Subpopulation (Witzell et al., *in review*), also support the increasing population hypothesis. Apparent increases in nesting in the Florida panhandle area are confounded by increases in monitoring, making interpretation uncertain.

Correlation analysis of the overlapping time series of available data (1989-1997) suggest that, in general, nest counts and strandings do not correlate well (Table 19). High correlation coefficients do occur between time series that are rapidly increasing (*e.g.*, Panhandle nests and northeast strandings), and these correlations disappear when the residuals are compared. The notable exception to the lack of correlation is between Northern nests and adults stranding in the Southeast Region ($r = 0.822$, $P=0.011$), a relationship that holds in the residuals of the ln-transformed data ($r = 0.822$, $P = 0.007$). Adult strandings in this region were also well correlated with nest counts in each of the 4 Northern Subpopulation States. Further analysis of strandings correlations by size were not possible because a much higher proportion of stranded turtles in other regions are reported without measurements (SE region reporting average is 52.5%, range 45-59%). Southeast strandings and Gulf of Mexico strandings were well correlated ($r=0.762$, $P=0.017$) but failed to correlate with northeast strandings. St. Lucie captures of adults and immatures correlated fairly well with South Florida nests and Southeast strandings, although the correlations were not significant at the $P<0.05$ level (raw data from the St. Lucie captures also correlated with Panhandle nests, but the residuals did not correlate). As more years of data are collected, more detailed time series analyses may reveal which sources of loggerhead population data correlate through time and are likely to be reasonable proxies for population size distributions and trends.

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Table 10. Natal origin of foraging-ground loggerhead sea turtles. Table values are proportional contributions.

Foraging Area	**Life History Stage(s)**	**N**	**Nesting Subpopulation**					
			Western North Atlantic				**South Atlantic**	**Mediterranean**
			N.C. to Northeast Florida	South Florida	NW Florida	Yucatán, Mexico	Brazil	Greece
Northeast U.S.[1, 13]	Primarily Benthic Immature	82	0.25	0.59	0.00	0.16	0.00	0.00
Chesapeake Bay*[2]	Primarily Benthic Immature	63	0.54	0.46				
Pamlico and Core Sounds, N. C.[3]	Benthic Immature	97	0.32	0.64	<0.01	0.03	0.01	0.00
S. Carolina*[4]	Benthic Immature	33	0.50	0.50				
Georgia*[5]	Benthic Immature	97	0.59	0.41				
Georgia[6]	Benthic Immature	192	0.24	0.73		0.03	0.00	0.00
Hutchinson Island, Florida[12]	Benthic Immature	109	0.10	0.69		0.20		
Florida Bay[7]	Benthic Immature and Adult	51	0.08	0.84		0.08		
Mediterranean Sea*[8,9]	Unknown	59		0.57				0.43
Azores and Madeira[10]	Pelagic Immature	183	0.19	0.71	0.00	0.11	0.00	0.00
Western Mediterranean[11]	Pelagic Immature	59	0.02	0.45				0.53
Eastern Mediterranean[11]	Pelagic Immature	52	0.02	0.47				0.51
Eastern Mediterranean[11]	Benthic Immature and Adult	58						1.00**

* Studies were conducted before full complement of genetic analyses of Atlantic nesting beaches were available.

** Includes rookeries of Turkey and Cyprus, in addition to Greece.

[1] Rankin-Baransky, 1997; [2] Norrgard, 1995; [3] Bass *et al.*, 1998; [4] Sears *et al.*, 1995; [5] Sears, 1994; [6] Bass *et al.*, 1999; [7] B. Schroeder, unpublished data; [8] Laurent *et al.*, 1993; [9] Bowen, 1995, [10] Bolten *et al.*, 1998, [11] Laurent *et al.*, 1998, [12] Witzell *et al.*, *in review*; [13] Rankin-Baransky *et al.*, *in review*.

Table 11. Loggerhead sea turtle nests in the U.S., 1989-1998. 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 (1993, 1994), Maley and Harris (1991, 1992), 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). Nesting female estimates are based on an average of 4.1 nests/female in all subpopulations.

Area	Year										Mean 1989-1998
	1989	1990	1991	1992	1993	1994	1995	1996	1997	1998	
Northern Nesting Subpopulation:											
North Carolina	457	806	931	729	485	1021	662	776	568	861	730
South Carolina	2444	4491	3657	3943	2757	4136	2959	3892	2169	4265	3471
Georgia	691	1085	1209	1054	475	1375	1028	1111	816	1067	991
N.E. Florida	778	1355	952	759	809	1355	1149	1142	893	1359	1055
Total	4370	7737	6749	6485	4526	7887	5798	6921	4446	7552	6247
Nesting Females	1066	1887	1646	1582	1104	1924	1414	1688	1084	1842	1524
South Florida Nesting Subpopulation:											
S.E. Florida	46295	62071	63416	59677	50618	64410	71394	68174	56266	74988	61731
S.W. Florida	2236	3085	3959	4186	3836	5395	6973	6461	6746	8454	5133
Total	48531	65156	67375	63863	54454	69805	78367	74635	63012	83442	66864
Nesting Females	11837	15892	16433	15576	13281	17026	19114	18204	15369	20352	16308
Florida Panhandle Nesting Subpopulation:											
Total	113	174	287	351	560	772	928	891	1133	1188	640
Nesting Females	28	42	70	86	137	188	226	217	276	290	156
Total U.S. Nests	53014	73067	74411	70699	59540	78464	85093	82447	68591	92182	73751
Total U.S. Nesting Females	12930	17821	18149	17244	14522	19138	20754	20109	16730	22483	17988

Table 12. Mean annual percent change in loggerhead nest numbers in the U.S. 1989-1998, based on slopes calculated from ln-transformed data. Bold F statistics are significant at $P < 0.05$; variance in nest counts from year-to-year and possible cyclical nesting reduced the power to detect trends for many areas. Data are for calculations of minimum number of nests and may not be accurate for assessments of trends due to yearly changes in nesting beach survey effort. See text for a discussion of trends in loggerhead nesting numbers for each subpopulation.

	North Carolina	South Carolina	Georgia	N.E. Florida	Total Northern		S.E. Florida	S.W. Florida	Total S. Florida	Florida Panhandle		Total U.S.
Percent change per year 1989-1998	1.7%	-0.1%	1.5%	2.9%	0.8%		2.9%	**13.2%**	**3.6%**	**2.6%**		**3.6%**
SE of slope	0.031	0.030	0.036	0.026	0.027		0.014	0.014	0.014	0.025		0.015
r-square	0.036	0.000	0.021	0.136	0.012		0.344	0.919	0.461	0.932		0.424
F	0.297	0.002	0.175	1.261	0.095		4.197	**90.805**	**6.840**	**109.060**		**5.895**

Table 13. Correlations (Pearson) of loggerhead nests from index beaches at Cape Island, Cape Romain National Wildlife Refuge, S.C., and Hutchinson Island, Fla. with nest totals (excluding each index beach) from each state and region, 1989-1998. Bold correlations are significant at $P < 0.05$. A. Raw nest numbers. B. Residuals from ln-transformed regression analysis. Raw nest numbers from Cape Island, S.C. and Hutchinson Island, Fla. were not significantly correlated ($r = 0.4667$, $P = 0.174$), but their regression residuals were highly correlated ($r = 0.7602$, $P = 0.01$).

	North Carolina	South Carolina (excluding Cape Is.)	Georgia	NE Florida	Northern Subpopulation (excluding Cape Is.)	SE Florida (excluding Hutchinson Is.)	SW Florida	S. Florida Subpopulation (excluding Hutchinson Is.)	Florida Panhandle
A. Raw nest numbers									
Cape Island, S.C.	**0.8132**	**0.8274**	**0.8412**	**0.6966**	**0.9017**	0.5615	-0.0251	0.4629	-0.2197
Hutchinson Island, Fla.	0.5307	0.5107	0.5750	**0.6511**	0.6213	**0.9547**	**0.7225**	**0.9456**	0.5389
B. Regression residuals									
Cape Island, S.C.	**0.8898**	**0.8247**	**0.9047**	**0.7809**	**0.9470**	**0.8813**	0.6312	**0.8719**	-0.0769
Hutchinson Island, Fla.	**0.6501**	**0.6470**	**0.6663**	0.6281	**0.7305**	**0.9414**	**0.8206**	**0.9395**	0.1378

Table 14. Correlations (Pearson) of U. S. loggerhead nests from all states and regions, 1989-1998. Bold correlations are significant at $P < 0.05$. A. Raw nest numbers. B. Residuals from ln-transformed regression analysis.

	North Carolina	*South Carolina*	*Georgia*	*N.E. Florida*	*Northern Total*	*S.E. Florida*	*S.W. Florida*	*S. Florida Total*	*Florida Panhandle*
A. Raw nest numbers									
North Carolina	1.000								
South Carolina	**0.818**	1.000							
Georgia	**0.935**	**0.733**	1.000						
N.E. Florida	**0.694**	**0.683**	**0.649**	1.000					
Northern Total	**0.920**	**0.963**	**0.865**	**0.799**	1.000				
S.E. Florida	**0.688**	0.617	**0.702**	**0.742**	**0.723**	1.000			
S.W. Florida	0.271	0.101	0.285	0.459	0.232	**0.758**	1.000		
S. Florida Total	**0.635**	0.543	**0.650**	**0.717**	**0.658**	**0.992**	**0.832**	1.000	
Florida Panhandle	0.103	-0.091	0.096	0.354	0.042	0.570	**0.960**	**0.665**	1.000
B. Regression residuals									
North Carolina	1.000								
South Carolina	**0.855**	1.000							
Georgia	**0.905**	**0.706**	1.000						
N.E. Florida	**0.695**	**0.693**	**0.633**	1.000					
Northern Total	**0.940**	**0.967**	**0.844**	**0.787**	1.000				
S.E. Florida	**0.828**	**0.807**	**0.822**	**0.722**	**0.873**	1.000			
S.W. Florida	**0.709**	0.525	**0.758**	0.431	**0.635**	**0.863**	1.000		
S. Florida Total	**0.824**	**0.790**	**0.826**	**0.713**	**0.863**	**0.999**	**0.876**	1.000	
Florida Panhandle	0.103	0.012	0.029	-0.037	0.040	0.108	0.447	0.115	1.000

Table 15. Loggerhead strandings by region.

	1986	1987	1988	1989	1990	1991	1992	1993	1994	1995	1996	1997	Total
N.E.U.S. Atlantic	93	175	166	134	207	158	206	221	227	299	274	346	2506
S.E.U.S. Atlantic	828	1179	904	896	1091	606	714	558	777	859	1127	902	10441
Gulf of Mexico	278	371	298	392	288	199	176	186	302	245	415	376	3526
Total	1199	1725	1368	1422	1586	963	1096	965	1306	1403	1816	1624	16473

Table 16. Loggerhead stranding trends. The annual rate of change for each region is based on a regression slope of ln-transformed strandings, with the model fit (r-square) given in parentheses. Slope values in bold are significant at $P < 0.05$.

				Annual percent change		
	1986-88	1995-97	Percent change	1986-1997 (r-square)	1989-1997 (r-square)	1991-1997 (r-square)
N.E.U.S. Atlantic	434 (10%)	919 (19%)	+ 112%	**+8.9%** (0.801)	**+10.0%** (0.846)	**+11.5** (0.911)
S.E.U.S. Atlantic	2911 (68%)	2888 (60%)	- 1%	-0.8% (0.017)	+1.5% (0.030)	**+9.1%** (0.659)
Gulf of Mexico	947 (22%)	1036 (21%)	+ 9%	0.0% (0.000)	+3.1% (0.068)	**+13.9%** (0.759)
Total	4292	4843	+ 13%	0.0% (0.017)	+3.1% (0.138)	**+10.5%** (0.835)

Table 17. Trends in loggerhead abundance at the St. Lucie Nuclear Power Plant, St. Lucie, Florida (Quantum Resources, 1995, unpublished data), over various time intervals to compare with nests and strandings. Bold values are significant at the P<0.05 level.

	Annual percent increase		
	1977-1998 (r-square)	1989-1998 (r-square)	1991-1998 (r-square)
All loggerheads	**4.2%** (0.401)	**14.0%** (0.783)	**17.0%** (0.764)
Adults (92 cm SCL)	**12.3%** (0.605)	11.2% (0.2282)	**25.3%** (0.546)
Large Benthic Immatures (70-91.9 cm SCL)	**5.5%** (0.377)	**10.1%** (0.467)	**17.0%** (0.687)
Small Benthic Immatures (<70 cm SCL)	**3.6%** (0.297)	**17.0%** (0.890)	**16.5%** (0.805)

Table 18. Observed size distributions for loggerheads in zones 29-32 of the southeast U.S., May 1997 - May 1998 (except January and February).

Data source	<70 cm SCL	70-91.9cm SCL	92cm SCL
Strandings Data (n=395)	77%	15%	17%
Naked Net Captures (n=20)[1]	92%	8%	0%

[1] Gulf & South Atlantic Fisheries Development Foundation, Inc. (1998), Jamir (1999)

Table 19. Pearson correlation coefficients between loggerhead nests and strandings in the southeastern U. S. and Gulf of Mexico and loggerhead captures at the St. Lucie Power Plant , 1989-1997. Bold coefficients are significant at P < 0.05. A. Raw catch data. B. Residuals from ln-transformed regression analysis.

	Northern nests	South Florida nests	Panhandle nests	NE US Strandings	SE US Strandings	Gulf of Mexico Strandings	SE US Adult Strandings	SE US Immature Strandings	St. Lucie all	St. Lucie adults
A. Raw data										
Northern nests	1.000									
S. Florida nests	0.604	1.000								
Panhandle nests	-0.146	0.542	1.000							
NE US Strandings	-0.128	0.530	**0.933**	1.000						
SE US Strandings	0.263	0.294	0.166	0.311	1.000					
Gulf of Mexico Strandings	-0.121	-0.026	0.300	0.261	**0.762**	1.000				
SE US Adult Strandings	**0.790**	0.549	0.219	0.250	0.592	0.310	1.000			
SE US Immature Strandings	0.268	0.186	0.041	0.220	**0.977**	**0.727**	0.556	1.000		
St. Lucie All	0.072	0.645	**0.735**	0.658	0.501	0.464	0.412	0.334	1.000	
St. Lucie Adults	0.089	0.458	0.399	0.370	0.655	0.522	0.433	0.518	**0.886**	1.000
St. Lucie Immatures	0.065	0.665	**0.785**	**0.701**	0.458	0.442	0.395	0.289	**0.995**	**0.836**
B. Ln-transformed residuals										
Northern nests	1.000									
S. Florida nests	**0.855**	1.000								
Panhandle nests	0.285	0.322	1.000							
NE US Strandings	0.126	0.218	-0.048	1.000						
SE US Strandings	0.251	0.223	**-0.728**	0.366	1.000					
Gulf of Mexico Strandings	-0.138	-0.256	**-0.824**	-0.165	**0.766**	1.000				
SE US Adult Strandings	**0.822**	0.534	0.009	0.286	0.556	0.237	1.000			
SE US Immature Strandings	0.237	0.194	**-0.761**	0.385	**0.988**	**0.775**	0.541	1.000		
St. Lucie All	0.193	0.319	-0.182	-0.068	0.532	0.393	0.330	0.430	1.000	
St. Lucie Adults	0.005	0.068	-0.275	0.202	0.633	0.509	0.369	0.555	**0.885**	1.000
St. Lucie Immatures	0.249	0.389	-0.118	-0.172	0.450	0.316	0.294	0.341	**0.984**	**0.790**

Figure 15. Loggerhead nesting areas, Western North Atlantic.

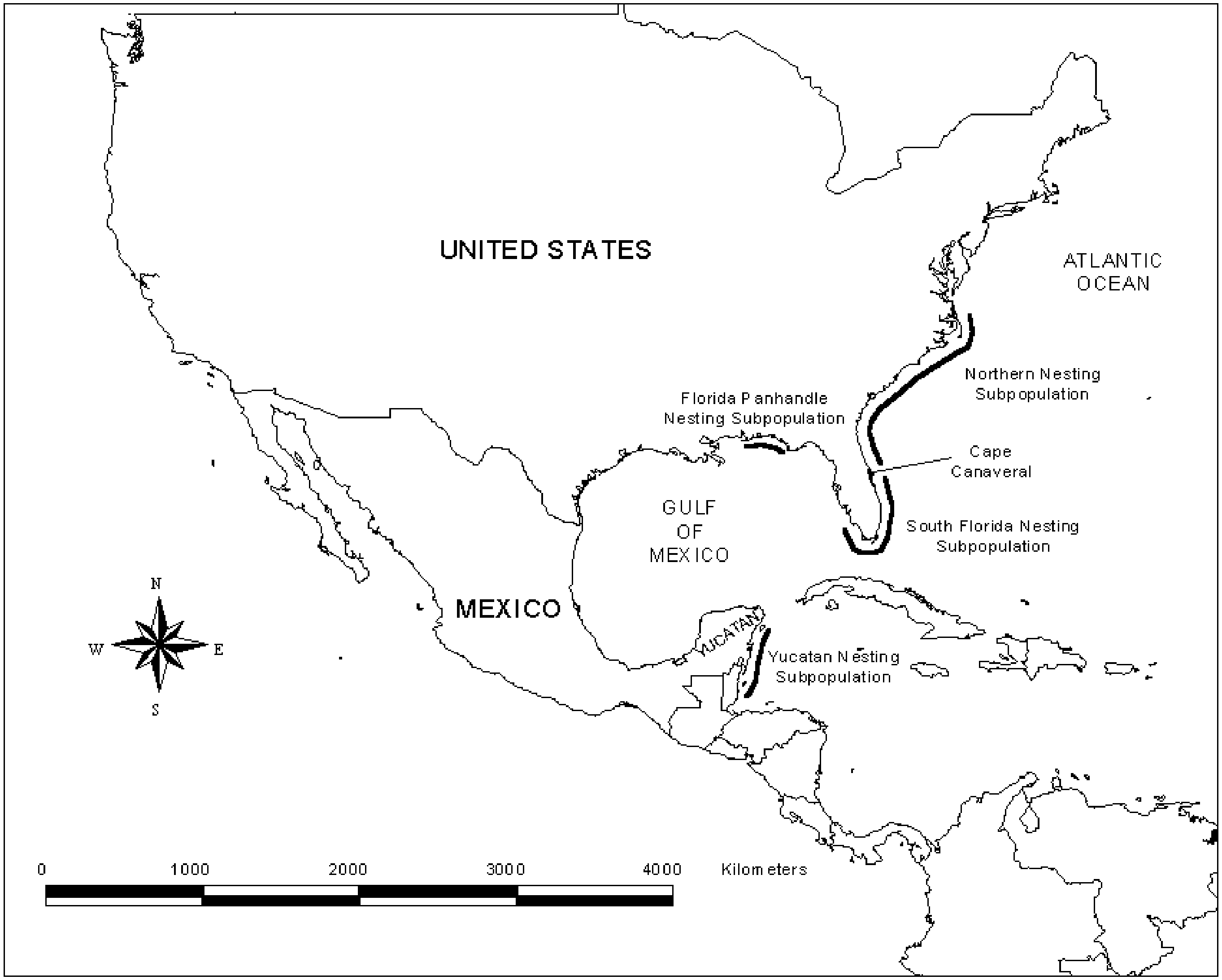


Figure 16. Loggerhead nests for the Northern Nesting Subpopulation, 1989-1998. A. Nest counts by state. B. Long-term nesting records for Cape Island, S.C.

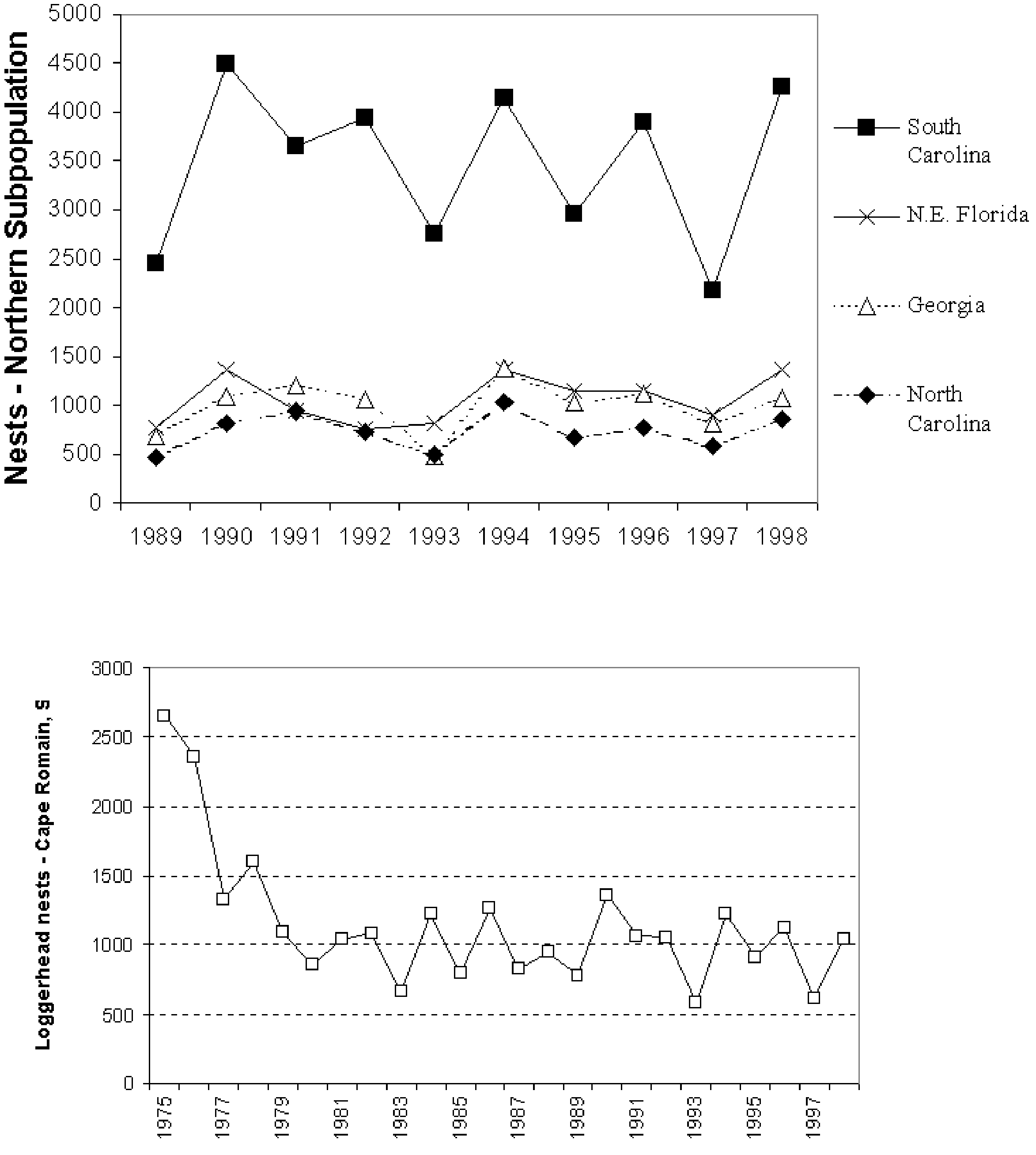
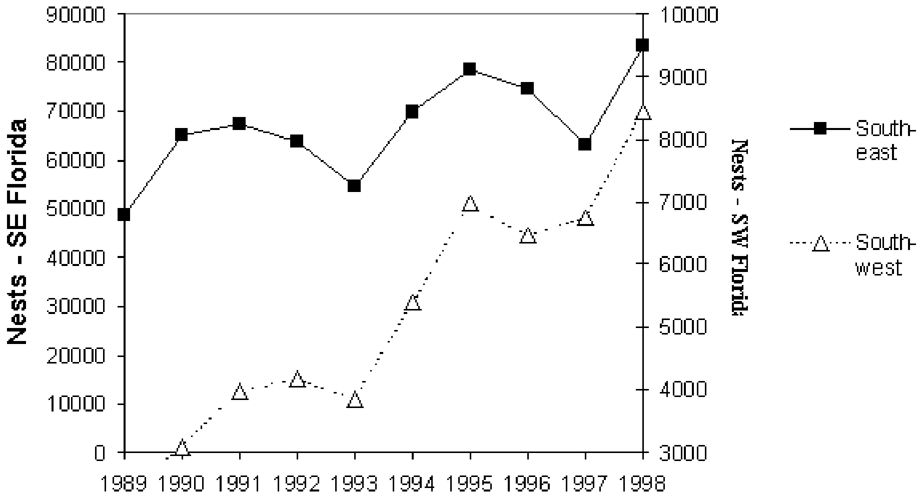


Figure 17. Loggerhead nests for the South Florida Nesting Subpopulation, 1989-1998. A. Nest counts by region. B. Long-term nesting records for Hutchinson Island, Fla.

A.



B.

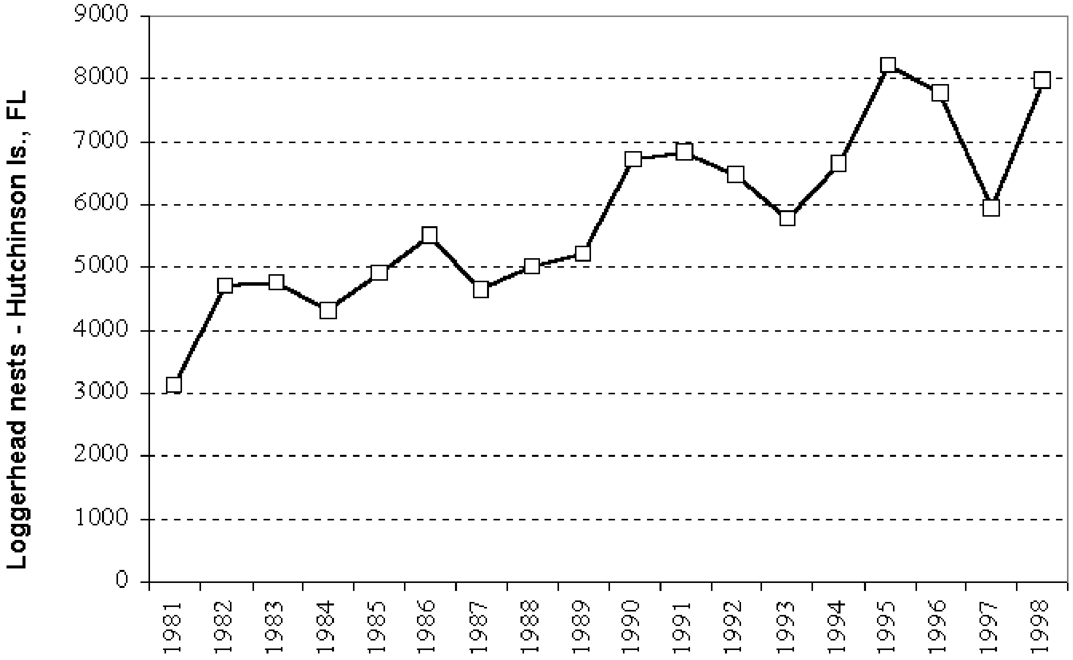


Figure 18. Loggerhead turtles captured by the cooling water intake pipe at the St. Lucie Nuclear Power Plant, St. Lucie, Fla., 1977-1998.

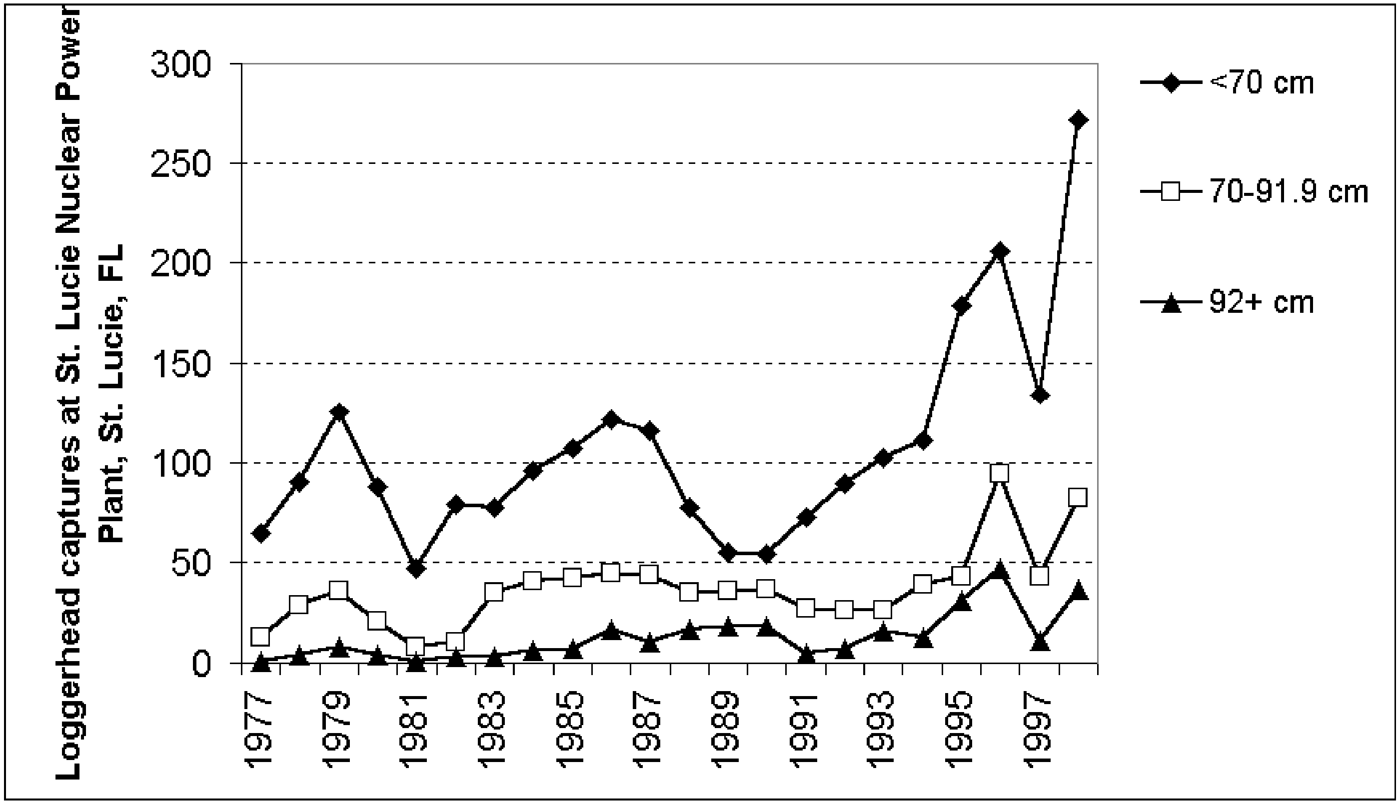


Figure 19. Size distributions and cumulative frequencies for loggerhead turtle strandings. A. Pooled data for 1986-1987. B. Pooled data for 1996-1997.

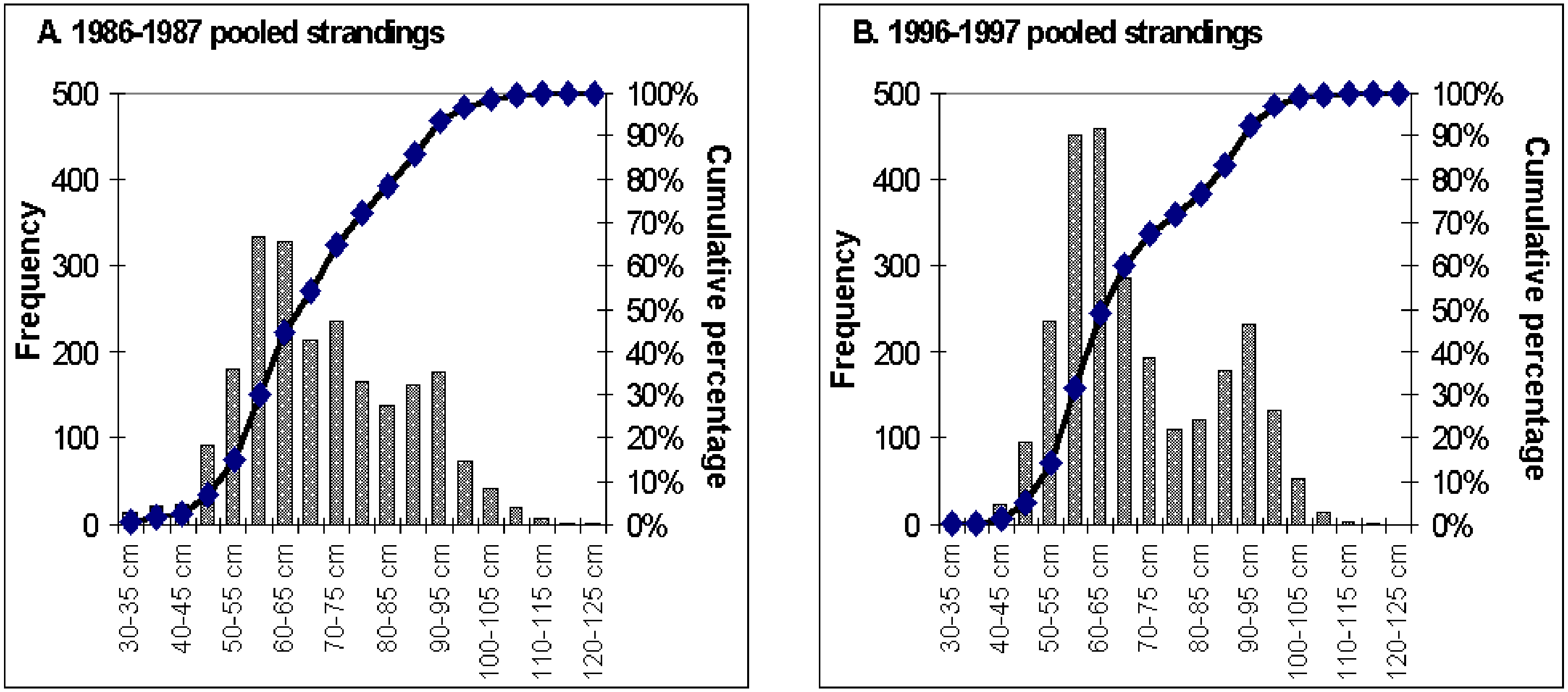
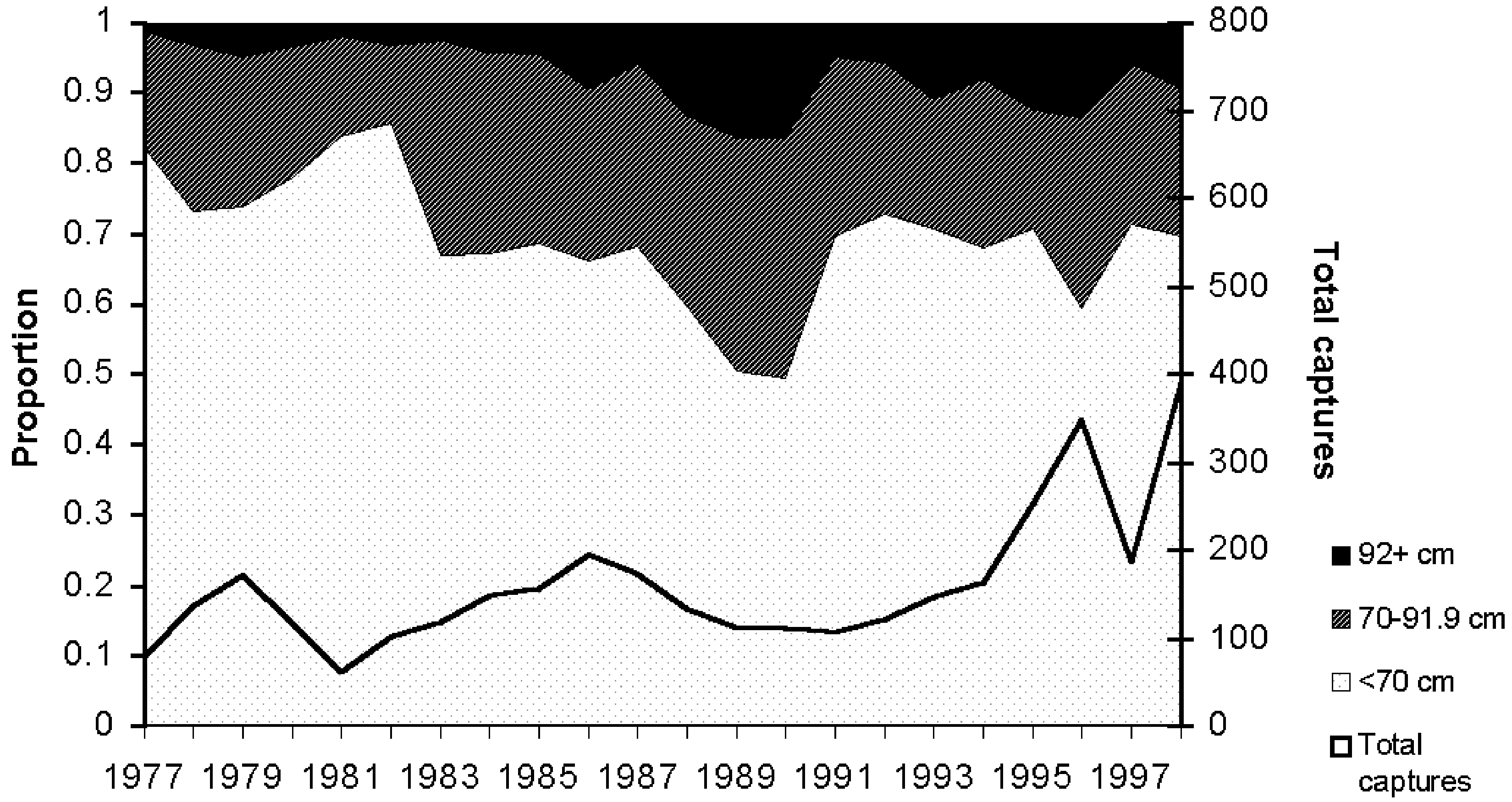


Figure 20. Size distributions of loggerheads captured at the St. Lucie Nuclear Power Plant, 1977-1998, represented as proportion of total captures. Time series of total captures is included (right y-axis).



Part 3. Anthropogenic Mortality Sources

Summary

A number of anthropogenic mortality sources have been identified for Kemp's ridley and loggerhead sea turtles (National Research Council, 1990; NMFS & USFWS, 1991; USFWS and NMFS, 1992) but few, outside drowning in bottom trawls, have been quantified with any degree of confidence. While they still cannot be quantified, new information in recent years leads the TEWG to be particularly concerned about longline fisheries and coastal gillnet fisheries, and about marine debris and pollution, mortality sources that primarily affect the pelagic immature stage. A more thorough assessment of anthropogenic mortality sources is provided in the first TEWG report (1998).

Mortality Sources for Pelagic Stage

Important sources of mortality occur in the pelagic phase of loggerhead turtles and are not revealed by numbers of stranded loggerheads. Likewise, this pelagic mortality is not accounted for in estimates of population size and mortality presented in TEWG (1998). This pelagic phase is approximately a decade in length (Bjorndal *et al.*, 2000; Bolten *et al.*, 1994).

Pelagic longline fisheries

The loggerhead model (Crouse *et al.* 1987; Crowder *et al.* 1994) indicated that the pelagic juvenile stage is likely the second most sensitive stage with respect to population growth rates, after the benthic juvenile stage. Pelagic loggerhead juveniles are believed to circumnavigate the North Atlantic, via the Atlantic Gyre, over a period of several years and have been reported from the Azores, and the Mediterranean as well as the eastern Caribbean. During this transit they are exposed sequentially to a series of longline fisheries that primarily target swordfish and tuna. Aguilar *et al.* (1995) estimated the Spanish swordfish longlining fleet in the Mediterranean, one of many fleets working in the region, alone captures more than 20,000 juvenile loggerheads (killing as many as 10,700) annually. About half of these are from southeastern U.S. beaches (Bowen *et al.,* 1993; Laurent *et al.,* 1993, 1998). Bolten *et al.* (1994, 1998) reported that virtually all of the pelagic immature loggerheads taken in the Azorean longline fleet are from western North Atlantic nesting subpopulations. In one study on the Mexican tuna longline fishery in the Gulf of Mexico, which included the capture of a Kemp's ridley, sea turtle mortality was 33% (Ulloa Ramirez and Gonzáles Ania, 2000), comparable to the mortality rates reported by Aguilar *et al.* (1995). Estimated bycatch of marine turtles by the U.S. Atlantic tuna and swordfish longline fisheries, based on observer data, was significantly greater than reported in logbooks (Johnson *et al.*, 1999; Witzell, 1999). Observer records indicated that an estimated 4808 loggerheads and no Kemp's ridleys were captured by the U.S. fleet 1992-1997 of which 21 were released dead (Johnson *et al.*, 1999). Logbooks indicated that loggerheads readily ingested hooks (Witzell, 1999).

Marine debris

An additional source of mortality that has not been adequately assessed is the ingestion of anthropogenic debris by pelagic turtles. Preliminary indications are that approximately 15% of pelagic post-hatchling loggerheads from Florida beaches have ingested plastics and approximately 46% have ingested tar within the first few weeks of pelagic foraging (n=168) (Witherington 1994, *in review*).

Mortality Sources for Benthic Feeding Stages

Trawls

Pursuant to Endangered Species Act Amendments of 1988, the National Research Council Committee on Sea Turtle Conservation assessed the status of the U.S. sea turtles (National Research Council, 1990). 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 500-5000 benthic immature and adult Kemp's ridley annual mortalities and an estimated 5,000-50,000 loggerhead mortalities were attributed to shrimp trawling (*Ibid.*), prior to implementation of TED regulations in 1989. This compares to 75-750 and 500-5000 estimated mortalities due to all other known human causes, respectively. The Council identified other fisheries, particularly the winter trawl fishery for summer flounder off North Carolina and Virginia as a possible source of turtle losses and estimated those as 50-200 per year. Among the findings of the National Research Council Committee on Sea Turtle Conservation (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, strandings 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; and

(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.

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, 28 June 1987). However, TEDs were first required in shrimp trawls beginning in May 1989, but were used only sporadically until May 1990, and then only seasonally (McDonald, 1990; Crouse *et al.*, 1992; Henwood *et al.*, 1992; Crouse, 1993; Weber *et al.*, 1995). Full implementation of seasonal TED requirements became effective essentially in May 1991. Beginning in December 1992, year-round use of TEDs was required by most trawlers operating in

Southeastern U.S. waters (57 FR p. 48861, Sept. 8, 1992) and this regulation was fully implemented by December 1994 (57 FR p. 18446, Apr. 30, 1992; 57 FR p. 57348, Dec. 4, 1992). In the three years before Federal Regulations (1986-1988), U.S. sea turtle strandings averaged 33% higher than in TED years 1991-1993 (Crouse, Crowder and Heppell *unpubl.* as cited by Crowder *et al.*, 1995).

The halt in the decline in numbers of nests at Rancho Nuevo can be attributed to a combination of factors, including a decrease in shrimp fishing effort in México waters when the U.S. shrimping fleet was prohibited from shrimping there in the late 1970s, and the overall decline and deterioration of the Mexican shrimping fleet. Threats to mating and nesting adult Kemp's ridleys from shrimping off Rancho Nuevo were reportedly high until the U.S. and México governments negotiated a bilateral agreement in 1976. Under this agreement U.S. shrimping in México waters was phased out through 1979 (Iversen *et al.*, 1993). After 1979, U.S. vessels continued to shrimp off México, although illegally and at reduced levels, through the mid-1980s, when the U.S. enforced the Lacey Act.

The reduction in shrimping effort caused by the departure of the U.S. fleet from Western Gulf waters of México was furthered by the subsequent decline and deterioration of the Mexican fleet. In 1995, the Mexican shrimp fleet in the Gulf was approximately 660 vessels, many of which were not actually fishing. There has been a decline in the number of vessels operating in the Campeche fleet, which currently represents about half of the operating México vessels, but the Tamaulipas fleet appears to be increasing (Table 20). 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 observing 10-20 trawlers operating 2-5 mi off the beach at night. Since 1995, Mexican waters of the Gulf of Mexico are closed to shrimping in concert with the Texas Closure, from May 15 through July 15, through binational agreement.

A study of the Mid-Atlantic winter trawl fishery for summer flounder identified the fishery as a significant source of sea turtle mortality, estimating that 1063 turtles, mostly Kemp's ridleys and loggerheads, were caught in a 4 month period and that 89-191 were killed (Epperly *et al.*, 1995, 1996). Consequently, since 1992, TEDs have been required in the fishery south of Cape Charles, Virginia. At first these regulations were implemented through emergency and interim rules and were seasonal requirements (57 FR p. 53603, Nov. 12, 1992; 57 FR p. 60135, Dec. 18, 1992; 58 FR p. 4088, Jan. 13, 1993; 58 FR p. 8554, Feb. 16, 1993). They became year round requirements (58 FR p. 48797, Sep. 20, 1993; 59 FR p. 19584, Mar. 7, 1994), and subsequently were codified in a final rule (61 FR p. 1846, Jan. 24, 1996) requiring TEDs in all summer flounder trawls operating south of Cape Charles, Va. There is a temporary reprieve of the regulation automatically occurring each year north of Oregon Inlet from Jan. 14-Mar. 15, when waters predictably are very cold.

Gill nets

Turtles are particularly susceptible to entanglement and drowning in gill net gear, especially when nets are set and left unattended. There are a number of gill net fisheries currently operating along the mid- and southeast U.S. Atlantic coastline that are known to incidentally capture loggerhead turtles. Although all or most nearshore gill netting in state waters of South Carolina, Georgia, Florida, Louisiana, and Texas is prohibited by state regulations, gill netting in other states' waters and in federal waters does occur. Of particular concern are the nearshore and inshore gill net fisheries of the mid-Atlantic operating in New Jersey, Delaware, Maryland, Virginia, and North Carolina state waters and/or federal waters offshore thereof. Annual peaks in loggerhead strandings on ocean beaches and just inside the mouths of bays and inlets regularly occur in early summer and late fall, coinciding with increased gill netting activity and observers have documented lethal takes of both loggerhead and Kemp's ridley turtles in these fisheries (D. Christensen, *personal communication*). In December 1999 the large mesh gill net fishery for southern founder in Pamlico Sound, N.C. was closed to fishing for 30 days, based on elevated levels of stranded Kemp's and loggerhead turtles in inshore waters and the documentation of takes by the fishery (64 FR, p. 70196, Dec. 16, 1999). On May 12, 2000 the large mesh gill net fishery along eastern North Carolina and Virginia, north of Cape Hatteras, including the ocean, was closed for 30 days because of elevated strandings off northern North Carolina, mostly loggerhead turtles, observers' documentation that the large mesh gill net fisheries for monkfish and dogfish do take turtles (D. Christensen, *personal communication*), and the fact that 4 of the stranded turtles were entangled in large mesh gill nets consistent with these fisheries (65 FR p. 31500, May 18, 2000). NMFS made a determination that the level of mortality was "severely impacting the northern nesting subpopulation of loggerheads", the first time management has identified a subpopulation of sea turtles as a management unit. Additionally, the gill net fishery targeting sharks in federal waters offshore Georgia and Florida warrants close observation as these fisheries are known to take loggerheads (Trent *et al.*, 1997) and likely could take Kemp's ridleys. Observers are required in this fishery during certain times of the year; however, it is unclear whether periods of expected highest turtle abundance are adequately covered.

NMFS has initiated discussions with coastal states of the mid-Atlantic in an attempt to address these take issues, however, little progress has been made (T. Conant, *personal communication*). There is concern by vessel captains over liability issues, and many have refused to take observers on board (S. Epperly, *personal communication*) despite the fact that many are classified as Category 2 fisheries under the Marine Mammal Act and are required to take observers if asked.

Hook and line

Loggerhead and Kemp's ridleys are known to bite a baited hook, frequently ingesting the hook. Hooked turtles have been reported by the public fishing from boats, piers, the beach, banks, and jetties (Cannon *et al.*, 1994; J. Braun McNeill, *personal communication*; A. Cannon *personal communication*; S. Epperly, *personal communication*) and from commercial fishermen fishing for reef fish and for sharks with both single rigs and bottom longlines (S. Epperly, *personal communicaton*). Necropsies of turtles have revealed hooks internally (Witzell and Teas, 1994), which often were the cause of death (W. Teas, *personal communication*). An investigation of injuries and mortalities related to fish hook ingestion is underway at the National MarineFisheries Laboratory in Galveston, Texas (T. Fontaine, *personal communication*) and NMFS currently is exploring adding questions about encounters with sea turtles to intercept interviews of recreational fishermen conducted by the Texas Parks and Wildlife Department and under the auspicies of the Marine Recreational Fishery Statistics Surveys conducted throughout the Gulf of Mexico and along the Atlantic Coast (T. Fontaine, *personal communication*; T. Conant, *personal communication*). NMFS also is considering questioning recreational fishermen aboard headboats throughout the southeast U.S. Atlantic and the Gulf of Mexico to quantify their encounters with sea turtles, also (J. Braun McNeill, *personal communication*).

Power plants

Power plants are known to entrain loggerhead and Kemp's ridley sea turtles at the intake canals to their cooling systems. In some cases, the turtles may become trapped on intake screens and drown if they are unable to surface. The cumulative effect of such mortalities is unclear, but is believed currently to be relatively small. Some plants, such as the Brunswick Steam and Electric Plant in Southport, N.C., placed permanent diversion screens over the mouths of their intake canals to keep turtles away from the intake screens of the plant but this has not been 100% effective (W. Pollard, *personal communication*). Sea turtle entrainment has also been reported at four other power plants in eastern Florida (National Research Council, 1990) and recently at the Crystal River plant in western Florida. This latter facility is of particular concern given its proximity to aggregations of subadult Kemp's ridley turtles in the eastern Gulf of Mexico (Schmid and Barichivich, *submitted*). Sea turtle monitoring at the Crystal River Energy Complex increased substantially in 1998 and 40 live takes (37 were Kemp's ridleys) and 5 lethal takes (all Kemp's), were observed that year (NMFS, 1999), mainly from February-May. The turtles were subadults, with carapace lengths of 21-55 cm. Through March 1999, 4 subadult Kemp's ridleys were removed from the intake bars and released alive (D. Bruzek, *personal communication*).

In some cases the turtles pass unharmed through intake pipes into holding ponds, where they are then captured and released. The best documented situation of this type of interaction is the St. Lucie Nuclear Power Plant (Florida Power and Light Co.) where, in recent years, capture rates have exceeded 200 loggerheads annually and the annual capture of Kemp's ridleys is generally less than 6 (Quantum Resources, 1995; J. Gorham, *personal communication*).

Marine debris

Of 1710 turtles necropsied between 1980 and 1992, 197 (11.5%) had ingested debris, including plastic pieces and balloons (Witzell and Teas, 1994): a greater proportion of loggerheads were affected than were Kemp's ridleys, and in both species the percentage impacted by digested debris was highest in the Gulf of Mexico. Studies conducted in the western Gulf of Mexico reported debris ingestion rates of 51% for loggerheads and 34% for Kemp's ridleys (summarized in Witzell and Teas, 1994). Apparently Kemp's ridleys in the northeast U.S. consume little or no plastic debris (Sadove and Morreale, 1990; Burke *et al.*, 1994).

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Table 20. Total shrimping vessels and vessels operating in the Campeche and Tamaulipas fleets, 1982-1995.

	Campeche Fleet		**Tamaulipas Fleet**	
Year	Total	Operating	Total	Operating
1982	605	578		
1983	575	507		
1983	544	481		
1985	512	357		
1986	488	354		
1987	469	368	unknown	124
1988	450	324	unknown	116
1989	423	294	unknown	139
1990	412	259	unknown	134
1991	356	225	unknown	145
1992	398	293	unknown	169
1993	403	325	unknown	163
1994	347	268	unknown	154
1995	352	292	unknown	199

Part 4. Take Limits

Introduction

One of the initial charges to the TEWG was to determine maximum take levels for each species. This is the maximum number of turtles that can be removed by human activities (*i.e.*, above losses due to natural mortality) without jeopardizing recovery goals specified in each Species Recovery Plan. While allocation of take is a regulatory action clearly beyond the scope of the Group, a scientific assessment of the population-level effects of current and maximum allowable anthropogenic mortality to achieve recovery goals is needed. In general, it was agreed that in order to meet recovery goals, "take" should not increase at a rate equal to or greater than the population growth rate for either species. However, reliable indicators of take levels and population growth were generally unavailable. The Group determined early on that observer coverage on shrimp vessels was an inadequate measure of fishing mortality; less than 1% of shrimp vessel trips include observers and most observers are on offshore vessels that catch few turtles, and there are few observer programs designed to document turtle take by other fisheries. Thus, strandings became the Group's proxy for take. Because the following key information was contentious or lacking, no method for setting strandings limits was completely satisfactory to all Group members:

> *Population size.* We do not have a reliable estimate of the number of turtles that are susceptible to fishing mortality in a given year. While there is a fairly good estimate of nest numbers, many assumptions must be made to translate nests into adult females, adult males, and juveniles of both sexes. For loggerheads, an approach that utilized the ratios of males:females and adults:juveniles in the strandings records gave estimatef of 183,627 - 316,113 turtles greater than 40 cm SCL (Table 3; TEWG, 1998). Because we have a time series of cohort sizes for Kemp's ridleys through records of hatchlings released at Rancho Nuevo, a population model was developed to determine total population size annually (see Part I., above). However, the estimate is completely dependent on unknown mortality rates, and varies by 10-fold. Both methods of estimating population size suffer from a lack of corroborating empirical estimates – we do not even know if we are within the correct order of magnitude.
>
> *Proportion of total mortality represented by strandings.* In order to estimate population size with the ratio method and to set take limits based on strandings, assumptions must be made about the relationship of strandings and the turtle population. There are two issues: whether the size distribution of stranded turtles is representative of the population as a whole, and what proportion of total mortality is represented by stranded turtles. The probability that a dead turtle will wash up on a beach is undoubtedly highly variable, depending on distance from shore, water temperature, wind and currents, and possibly body size. Fishing mortality may be size-selective and is also variable. The group agreed that a majority of turtles that are killed in fishing gear fail to strand, but an actual proportion could not be agreed upon because existing data are not adequate to evaluate this.

> *Population distribution.* Turtles of different sizes are not evenly distributed in time and space. Migrating turtles may be exposed to different levels of risk from fishing mortality as their overlap with fishing fleets varies through the year. Juvenile as well as adult loggerheads from genetically distinct subpopulations appear to overlap disproportionately in space, but the genetics of turtles caught by various fishing fleets has not been determined. This has profound implications for the ratio method of population size estimation and for setting take limits based on strandings.

Several other unknowns also impeded development of take limits, including age at maturity (to determine time lags). Any one of these data deficiencies could have been surmounted with careful analysis of the best information available; the lack of consensus about take limits primarily rose from the unknown consequences of "stacking" so many uncertainties. Setting a take limit or estimating the impact of take is necessarily a quantitative exercise that requires a substantial amount of information and a good understanding of how data uncertainty will affect the results (Mangel, 1993; Taylor *et al.,* 2000). Although three methods to set strandings limits were used at various times by the TEWG, each was presented as an Interim Stranding Limit (ISL) to be used for a single year, rather than an agreed-upon method to be incorporated into management.

Three Interim Strandings Limits (ISL) calculations: Methods and Limitations

1) Running average

In 1996 and 1997, Kemp's ridley ISLs were calculated according to a 5 year running average of strandings + 1 standard deviation, which was then multiplied by one-half the observed rate of increase in the nesting population. The observed rate of increase in nests was estimated as the slope of the natural log of nest levels over the past 5 years, transformed to a linear (proportional) scale (TEWG, 1998). Stranded Kemp's ridley turtles greater than 10 cm, including those from cold-stunning events, were used in these analyses. In 1997, the stranding data were partitioned according to geographic regions (western Gulf of Mexico, zones 13 - 21; eastern Gulf, zones 1-12 and partial zones 24 and 25; southeastern U.S. Atlantic, partial zones 24 and 25, 26-35, partial zone 36; northeastern U.S. Atlantic, partial zone 36, 37 - 44). Similar methods were used for calculating loggerhead ISLs in 1996 and 1997 with the exception of the nesting multiplier and the inclusion of cold-stunned turtles. During both years, stranded loggerhead turtles greater than 40 cm were partitioned among the four regions. In 1997, the standard deviation was not added to the running averages of loggerheads on the Atlantic coast in order to give more conservative ISL estimates. Furthermore, log-normal means were investigated for both species in 1997, but the NMFS Southeast Regional Office, when presented with two sets of ISLs, chose to continue with arithmetic means. In 1998 the Group agreed that the running average method was inadequate for calculating ISLs, primarily because this method does not account for changes in fishing mortality, growing concern about the status of the Northern Subpopulation of loggerheads, and the large number of loggerheads stranding and the size distribution of those animals.

2) Slope method

For Kemp's ridleys, it appeared that in many years the rate of increase of strandings exceeded the observed rate of increase of nests and hatchlings at Rancho Nuevo. A new method was introduced in 1998 that worked on the key assumption that to maintain and enhance population growth that rate of increase of strandings should be lower than the population rate of increase. This method relied on the estimated slopes of hatchling production and strandings and set the ISL for the following year to obtain a desired slope. It was used on a trial basis to calculate an ISL for Kemp's, as a full analysis of potential changes in population size and strandings on the ISLs had not been attempted. As a measure of population growth rate, the group agreed to use hatchling production 2 years prior to the strandings year. Best estimates from correlation studies suggest that turtles spend two years at sea (see Life Stage Definitions in the Kemp's Ridley Status Report, above). This is corroborated by the new catch curve analysis, which shows peak strandings age of 2-3 years old in most strandings years (see Parameterization in the modeling section of the Kemp's Ridley Status Report, above). The strandings years used in the analysis were 1986-1997 for all regions, and did not include headstarted, TED-tested, or cold-stunned turtles and also excluded post-hatchlings (< 10 cm).

Slope calculation

Hatchlings released 2 years prior and strandings were ln-transformed to calculate and compare slopes (Table 21). Slopes and regression statistics were calculated with Excel 5.0. Over the 12 year time period the two slopes were nearly identical ($m = 0.8$ for hatchlings and $m = 0.78$ for strandings). Strandings were much more variable than hatchlings, but the regression was still significant at the 0.05 level ($p = 0.028$). The standard error of the hatchling slope was 0.011.

Setting the ISL

Strandings in 1998 should not exceed a value that maintains or lowers the slope to a desired level. To set the ISL conservatively, the group agreed that the desired strandings slope should be 1 standard error below the slope observed for hatchling production. Using a minimization program in Excel (Solver), the number of allowable strandings in 1998 was calculated to give a strandings slope of 0.069 over the years 1986 - 1998 (0.069 = observed slope of 0.08 - 0.011, the standard error). That point estimate was 334 turtles (Figure 21).

Caveats of the slope method

The group agreed that this was an interim method in need of further research, but was a more appropriate and conservative estimate than the running average method. Several concerns were raised. First, the goal of the method must be determined: do we want to reduce the rate of strandings increase to assure that the population will continue to increase rapidly, or is the "conservative factor" of reducing the slope by 1 standard error unnecessary? Regardless of whether the average slope of the strandings is greater than the population slope, if the slopes are not statistically different from each other a reduction in the slope may not be needed. One problem with determining statistical differences between the two slopes is that their variances are not equal ($p = 0.35$, one-tailed F test), which make standard slope comparisons (t-tests) invalid. Second, setting the strandings limit for a single year may not be feasible if the desired slope is very different from the observed strandings slope. This will depend on the number of years used in the analysis; 12 years may be inappropriate for Kemp's because it includes hatchling production years when the population was declining and spans more than the current predicted generation time (10 years).

3) Potential Biological Removal (PBR)

Potential Biological Removal (PBR) was developed for marine mammal stocks (Barlow *et al.*, 1995), which share many life history traits and data restrictions with sea turtles. PBR is based on the concept that human-caused mortality of a protected mammal stock should not exceed one-half the potential net productivity rate of the population, adjusted by a recovery factor (F) that varies from 0 to 1. It is a simple equation that requires a minimum population size estimate (N_{min}), the maximum rate of increase predicted (or measured) for a population (R_{max}), and pre-determined risk criteria for the recovery factor:

$$PBR = N_{min} \times (0.5 \times R_{max}) \times F$$

Wade (1998) ran simulations for a number of marine mammals to determine appropriate recovery factors for endangered and threatened populations. The default F values are: widespread populations with little risk of depletion, $F = 1.0$, threatened species $F = 0.5$, and endangered species $F = 0.1$. It has been determined that a stock experiencing a level of human-caused mortality exceeding PBR could become depleted (Wade 1998). Although data do not exist to conduct comparable simulations for sea turtle stocks, there are enough life history similarities between the two taxa (relatively low fecundity, advanced age at maturity, longevity, etc.) that the majority of the TEWG felt the PBR concept could be applied to sea turtle stocks. Discussions with the developers of the marine mammal PBR methodology have suggested that with modification, such as calculating separate PBRs for each life stage, PBR is potentially applicable to sea turtles. Modifications should account for the large difference in the reproductive value and abundance of different life stages and the extreme delay in maturity for some species.

PBR calculations for sea turtles

The PBR estimate varies depending on each of the three required parameters. As an example, Figure 22 shows changes in PBR with various recovery factors when R_{max} is set at 0.12 (*i.e.*, observed rate of increase for Kemp's ridleys, 1985-1998) or 0.04 (observed rate of increase for loggerhead nests in South Florida, 1989-1998). Note that these rates of increase are probably below R_{max} because of incidental mortality. For a population of 2500 turtles, total human-induced take would range from 10 to 1000 individuals. The lowest PBRs occur for very slow-growing populations, where additions to the population (recruitment) each year are only a small proportion of the total population size. While PBR numbers can be very low if conservative recovery factors are used, it is important to realize that PBR represents a conservative estimate of allowable take that will not prevent population recovery or cause a population decline (Wade, 1998). Also, if PBR is used to determine ISLs, additional multipliers are required to reduce total take by the proportion of animals killed that are likely to be counted as strandings.

Application to adult loggerheads

Because loggerheads have such a long lag time between hatching and maturation, the slope and running average methods were judged to be inadequate for protection of the threatened Northern Subpopulation by a majority of the Group in 1998. PBR was used to calculate the number of adult loggerhead strandings allowed in southern Atlantic fishing zones. Strandings for all of the zones within an area were used to determine sex ratios of adults. Also, nesting data were combined for the S. Florida and the Florida Panhandle subpopulations. PBR for adult loggerheads was calculated assuming the minimum population size was the estimated size of the adult populations in U.S. waters, the maximum rate of increase was 5%, and the recovery factor was 0.5, given the species is listed as threatened. It was assumed that strandings represent, at best, 25% of the at-sea nearshore mortality (Murphy and Hopkins-Murphy, 1989). Thus, the ISL was set as 25% of the calculated PBR for each area/subpopulation (Table 22) The result was a mean of 9 adult strandings in the Northern Subpopulation zones and 98 adult strandings in all remaining areas. In addition to the assumptions of the ratio method which are listed in TEWG (1998), this method assumes that adult individuals of the Northern Subpopulation are found in zones 29-44 only, or are outside U.S. waters. ISLs for turtles <92 cm SCL (and unmeasured) were calculated by region using the running average method.

Caveats of PBR

PBR has yet to be evaluated for sea turtles, contains a number of simplifying assumptions, and the default recovery factors are considered arbitrary by some researchers. A minority opinion to the ISL recommendations for 1998 strongly criticized the use of PBR and the emphasis on separating the loggerhead ISLs by subpopulation. None of the parameters in the PBR equation have been fully evaluated, including the strandings:deaths ratio required to set ISLs. Finally, PBR does not include much of the important biological information available for sea turtles, such as life history and sensitivity of various life stages.

Summary of results 1996 – 1998

Strandings of Kemp’s ridleys during 1996-1998 for comparison to the ISLs set were 393, 458, and 412, respectively (Table 23). Strandings of Kemp’s ridleys in 1996 and 1997 were less than the ISLs set for those years. However, strandings of Kemp’s ridleys in 1998 (412) exceeded the ISL of 334 animals by nearly 25%.

Total strandings of loggerheads during 1996-1998 exceeded total ISLs by 12-82% each year (Table 23). In 1996 ISLs were exceeded in all regions except the northeast U.S. and in 1997 ISLs were exceeded in all regions. In 1998 ISLs for juvenile loggerheads were exceeded in all regions except the Eastern Gulf of Mexico. The number of adults stranding in zones 29-44, the area where all adults are presumed to be from the Northern Subpopulation, was extraordinary (83) exceeding

ISL (9) by 922%, even exceeding the number of adults stranding in other zones where they are presumed to be from the much larger (nearly 10x) S. Florida Subpopulation and from the Florida Panhandle Subpopulation.

Research recommendations for future evaluation of take limits

Take limits are a contentious issue because data on sea turtles are extremely limited and scientific opinions vary on the importance of conservative recovery factors. Better estimates of means and uncertainty in population size, the strandings:deaths ratio, and population distribution will help, but the level of allowable risk must be established first. As long as sea turtle populations appear to be recovering, it is unlikely that disagreements about risk aversion will be resolved. The distinctness of the loggerhead subpopulations is critical, as the nests of Northern Subpopulation beaches do not appear to be increasing. Because the goals stated in the Recovery Plan specify nest increases for these areas, thoughtful evaluation of management priorities is warranted.

A simulation analysis is underway to examine the effects of population change and variance, number of years analyzed, and parameter uncertainty in the ISLs calculated using the slope method, running averages, and PBR (S. Heppell, *personal communication*). It may also be possible to use the Kemp's ridley age-based population model for projections through time to see how the population might be affected by these take levels, but that will require an estimate of the strandings:total mortality proportion. A future meeting of population modelers familiar with a variety of methods for analyzing sea turtle and fishery data may also be beneficial, particularly if a working simulation model could be provided to test various take limits.

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Table 21. Hatchling production and strandings levels used to calculate slopes.

Year	Hatchlings produced two years earlier	Ln (hatchlings)	Strandings	Ln(strandings)
1986	58124	10.970	306	5.724
1987	51033	10.840	162	5.088
1988	48818	10.796	216	5.375
1989	44634	10.706	150	5.011
1990	62218	11.038	258	5.553
1991	66752	11.109	165	5.106
1992	74339	11.216	156	5.050
1993	76238	11.242	327	5.790
1994	92116	11.431	550	6.310
1995	84210	11.341	396	5.981
1996	107687	11.587	382	5.945
1997	120038	11.696	412	6.021
slope		0.080		0.078
standard error		0.011		0.030
p-value		<0.001		0.027
Lower 95% C.I.		0.056		0.010
Upper 95% C.I.		0.104		0.146

Table 22. PBR and ISL estimates for adult loggerheads, 1998.

NORTHERN SUBPOPULATION

	INPUT DATA									
		Stranded Animals >=92 cm, May-August								
Year	Number of Nests Z. 29-44	Females	Males	Unknowns	Est. Prop. Females	Est. No. of Nesting Females	Estimated Number of Adult Males	Est. Adult Pop. In U.S. Waters	PBR	ISL
1989	4372	3	5	17	0.46	1066	1252	2318	29	7
1990	7746	13	6	17	0.60	1889	1274	3163	40	10
1991	6749	10	6	19	0.56	1646	1308	2955	37	9
1992	6487	9	12	13	0.46	1582	1888	3471	43	11
1993	4528	6	13	17	0.40	1104	1638	2742	34	9
1994	7899	22	16	20	0.55	1927	1565	3492	44	11
1995	5798	18	15	28	0.52	1414	1282	2696	34	8
1996	6940	18	23	29	0.46	1693	1953	3646	46	11
1997	4220	18	22	16	0.46	1029	1188	2217	28	7
Mean	6082				0.50	1483	1498	2982	37	9

SOUTH FLORIDA SUBPOPULATION AND FLORIDA PANHANDLE SUBPOPULATION

	INPUT DATA									
		Stranded Animals >=92 cm, May-August								
Year	Number of Nests Z. 1-28	Females	Males	Unknowns	Est. Prop. Females	Est. No. of Nesting Females	Estimated Number of Adult Males	Est. Adult Pop. In U.S. Waters	PBR	ISL
1989	48644	8	13	15	0.43	11864	15692	27556	344	86
1990	65330	14	7	14	0.60	15934	10623	26557	332	83
1991	67662	11	9	15	0.53	16503	14719	31222	390	98
1992	64214	15	10	12	0.57	15662	11933	27595	345	86
1993	55012	12	16	10	0.45	13418	16575	29992	375	94
1994	70387	11	8	20	0.54	17168	14715	31883	399	100
1995	79508	6	12	5	0.37	19392	33081	52473	656	164
1996	75526	19	9	19	0.61	18421	11957	30378	380	95
1997	63193	12	13	11	0.49	15413	16294	31707	396	99
Mean	65497				0.51	15975	15454	31429	393	98

Table 23. Summary of Incidental Stranding Limits (ISL) and total strandings for 1996-98. ISL methods for loggerheads in 1998 are from the majority report.

Year	*Species*	Region (zones)	Size classes[1]	Cold-stunned included	Calculation method	ISL	Total strandings
1996	*L. kempii*	All	> 10 cm	yes	(Arithmetic mean + 1 std dev)*multiplier	586	393
	C. caretta	Western Gulf (13 - 21)	> 40 cm	no	Arithmetic mean + 1 std dev	164	206
		Eastern Gulf (1 - 12, part 24 & 25)	> 40 cm	no	Arithmetic mean + 1 std dev	127	204
		Southeast US coast (part 24 & 25 - 35, part 36)	> 40 cm	no	Arithmetic mean + 1 std dev	803	1131
		Northeast US coast (part 36, 37 - 44)	> 40 cm	no	Arithmetic mean + 1 std dev	269	244
1997[2]	*L. kempii*	Western Gulf	> 10 cm	yes	(Arithmetic mean + 1 std dev)*multiplier	311	228
		Eastern Gulf	> 10 cm	yes	(Arithmetic mean + 1 std dev)*multiplier	98	80
		Southeast US coast	> 10 cm	yes	(Arithmetic mean + 1 std dev)*multiplier	117	93
		Northeast US coast	> 10 cm	yes	(Arithmetic mean + 1 std dev)*multiplier	113	57
	C. caretta	Western Gulf	> 40 cm	no	Arithmetic mean + 1 std dev	192	170
		Eastern Gulf	> 40 cm	no	Arithmetic mean + 1 std dev	174	197
		Southeast US coast	> 40 cm	no	Arithmetic mean without std dev	792	892
		Northeast US coast	> 40 cm	no	Arithmetic mean without std dev	233	311
1998	*L. kempii*	All	> 10 cm	no	Slope method	334	412
	C. caretta adults	S. Fla & Panhandle subpop.s (1 - 28)	92 cm	no	PBR estimate	98	59
		Northern subpopulation (29 - 44)	92 cm	no	PBR estimate	9	83
	C. caretta juveniles	Western Gulf	40 - 91 cm & unmeasured	no	Arithmetic mean without std dev	129	170
		Eastern Gulf	40 - 91 cm & unmeasured	no	Arithmetic mean without std dev	131	123
		Southeast US coast	40 - 91 cm & unmeasured	no	Arithmetic mean - 2 std dev	380	904
		Northeast US coast	40 - 91 cm & unmeasured	no	Arithmetic mean - 2 std dev	97	343

1. Unmeasured turtles were included.

2. In 1997, methods based on both geometric and arithmetic 5-yr averages were used. However, the NMFS Southeast Regional Office chose to use ISLs based on the arithmetic means.

Figure 21. Slope method for ISL, showing strandings level required to achieve desired slope of m=0.069.

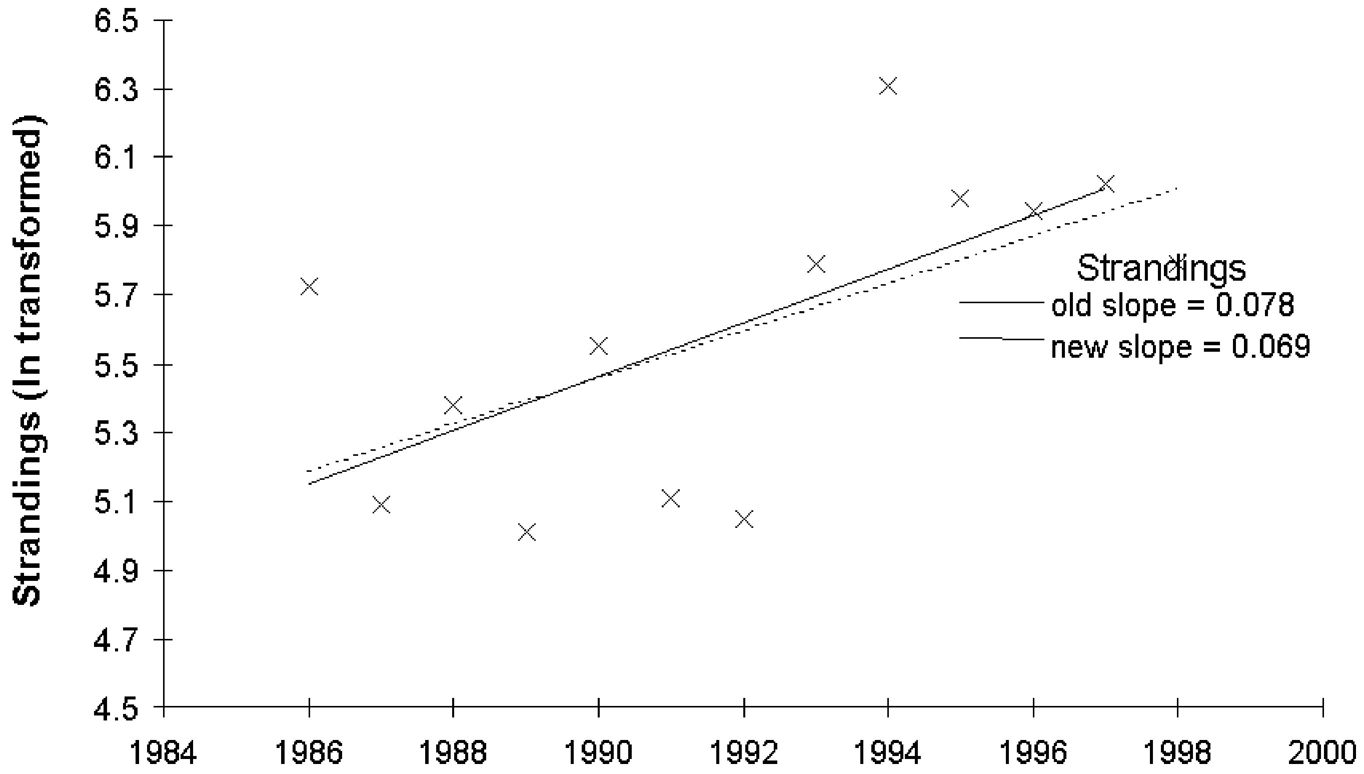
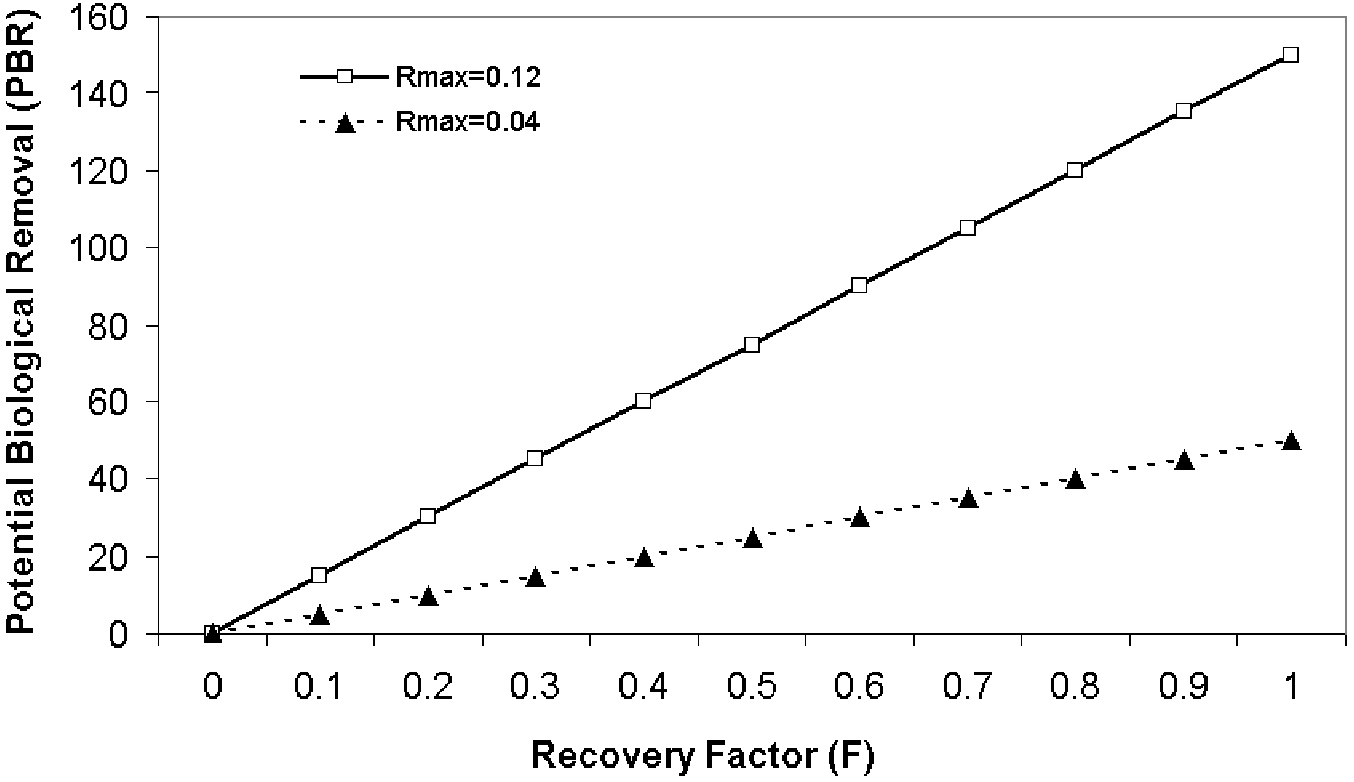


Figure 22. Effect of the recovery factor (F) on PBR for a population of 2500.



Part 5. Research Recommendations

Progress towards developing realistic population level models is dependent upon obtaining empirically derived parameters which reflect and incorporate the life history of each species. To this end, the information required includes empirically estimated survival rates for each life history stage defined by age. The assignment of ages and stages requires the application of age-length keys developed from growth models, which are used to develop the age structure for the population. The evaluation of changes in the age structure of the population can be completed by examining shifts in size frequency distributions over time and between areas. Shifts in abundance and the age structure must be evaluated in the context of migration in and out of areas identified as developmental or important foraging areas. This information can only come from research on turtles in the water and requires the accumulation of mark-recapture information over several years in multiple areas of known utilization. These in-water areas must be sampled to ensure that growth models, abundance indices, and size/age frequency distributions can be obtained. The distributions for these must be described with some measure of variability so that trends and shifts can be evaluated. Problems with data collected from strandings to estimate age and growth and survivorship have been demonstrated and underscored the need to conduct long term in-water studies that are comparable between years and between areas.

In-water surveys will provide data to evaluate changes in the age structure of populations. This trend analysis should enable us to evaluate progress towards recovery as defined in the individual recovery plans. Currently, data on the number of nesting females or nests is used to evaluate recovery and to ensure recovery. The timeline to the fully recruited benthic stage is shorter than that for returning to the nesting beach, especially for loggerhead turtles. Tracking recovery requires tracking juvenile turtles to ensure recovery rather than waiting the many years to maturity, which may be too late to reverse any declines.

Strandings data continue to provide information about unusually large mortality events that can be attributed to either a natural event or anthropogenic activity. These data also continue to provide information on sex ratios, maturity, breeding condition, and size frequencies. Strandings can be used to determine population demographics and to estimate mortality rates, but only if they are representative of the live turtle population. This assumption must be verified, as it continues to limit the interpretation of results that rely on these data. The relationship between the location and occurrence of strandings and environmental factors such as tides, currents, water temperature, resource condition and quantity must be determined to fully understand where the mortality occurred relative to where the stranding is reported. Stranding protocols have been recently standardized to provide comparable information between areas and time periods; accumulation of these data over several years may provide trend information if strandings are representative of the living population.

Research must continue to pursue ways to age turtles and to validate these ages. Body parts, such as the humerus, ear bone, or other hard structures that may harbor growth rings should be carefully preserved for all specimens collected by the Sea Turtle Stranding and Salvage Network, particularly body parts from known-age turtles. Validation of ages determined by humerus or sclera growth should be attempted through the use of hard part markers. The use of internal wire coded tags placed in small Kemp's ridleys should be continued to provide aging information. However, the success of this effort can only be ensured through the distribution of tag detectors throughout the range for this species and for use with both dead and live turtles.

While estimating total mortality rates empirically is an immediate and critical requirement, the partitioning of mortality by cause is imperative for the management of these species if recovery goals are not being met. When mortality can be partitioned and identified by cause, then management measures can be implemented to eliminate or reduce controllable anthropogenic sources of mortality. Interactions with fisheries gear can be fully evaluated and quantified with the placement of observers at a statistically valid sampling level to measure incidental take and mortality by this component on a fishery by fishery basis. There is no substitute for the direct observation of take and mortality as evidenced by the effort to quantify these in the commercial shrimp trawl fishery of the southeast U.S. Repeated mortality incidents occur where high levels of strandings are reported coincident with anecdotal observations of nearshore trawling activities, but the lack of observer coverage on nearshore vessels prevents verification and quantification of a causal relationship between trawling activity and subsequent strandings.

Genetic studies must be continued to fully understand and quantify the stock structure of the loggerhead turtle within U.S. Atlantic waters. Currently, the TEWG describes sub-populations based on nesting location as supported by recent genetic studies. However, these studies are limited to female turtles and do not provide any measure of fidelity by males. Foraging ground studies provide information on the mixing of subpopulations. The extent to which cross over by males occurs likely will determine the actual stock structure for this species. However, the TEWG notes that the natal homing of female turtles, as verified by existing genetic data, suggests that a nesting assemblage could be lost for ecological time periods if a subpopulation declines to extinction.

The pursuit of stock assessments is contingent upon these recommendations to enhance existing research programs and develop new ones that can determine the status and condition relative to recovery of these species. Limitations of the existing data and time series continue to be the critical impediment to completing these analyses. Fine-tuning of data that have not been collected for the purpose of providing empirically based population parameters will not further the efforts towards a stock assessment. Peer reviewed results are certainly preferable and ideal for inclusion in such stock assessments. It is the conclusion of the TEWG that further pursuit of stock assessment analyses is unwarranted until these data and results are available to the group.

Appendix 1. Von Bertalanffy Growth Models for Wild Kemp's Ridley Turtles: Analyses of the NMFS Miami Laboratory Tagging Database

Jeffrey R. Schmid and Amy Woodhead

National Marine Fisheries Service
Southeast Fisheries Science Center
75 Virginia Beach Drive
Miami, FL 33149

Prepared for
The Turtle Expert Working Group

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A number of von Bertalanffy growth models have been developed for the Kemp's ridley turtle, *Lepidochelys kempi*. Zug and Kalb (1989), Zug (1990), and Schmid (1995, 1998) prepared von Bertalanffy growth models for wild, subadult Kemp's ridley turtles, but the range of carapace lengths and sample sizes for these studies were insufficient to accurately describe growth parameters. Caillouet *et al.* (1995) provided a growth model for head-started Kemp's ridley turtles released off the Texas coast and subsequently recaptured in the Gulf of Mexico. Zug *et al.* (1997) recently refined their skeletochronological growth model by increasing the sample size and the size range of turtles. Schmid and Witzell (1997) provided von Bertalanffy equations for the combined data of tagging studies in Florida (Schmid, 1995, 1998) and subsequent recaptures at Ranch Nuevo, Mexico. This paper presents von Bertalanffy growth model analyses for Kemp's ridley turtles using the data available from the National Marine Fisheries Service (NMFS) Miami Laboratory Cooperative Marine Turtle Tagging Program.

Materials and Methods

Tagging records for wild Kemp's ridley turtles were compiled from several sources: NMFS Miami laboratory tagging database[1,2,3,4,5,6,7,8,9,10], Cape Canaveral from 1986 to 1991 (Schmid, 1995), and Cedar Keys from 1986 to 1995 (Schmid, 1998). Carapace measurements were converted to metric units, as necessary, and transformed to standard straight-line carapace length (SSCL; nuchal notch to posterior end of postcentral scute) using the regression equations of Teas (1993), Schmid (1998), and Schmid and Witzell (1997). Records for a NMFS Galveston headstart turtle, a negative growth rate (-5.6 cm/yr), and an exceedingly large growth rate (493.1 cm/yr) were omitted from the analyses. Means are followed by ± one standard deviation.

The von Bertalanffy growth interval equation (Fabens, 1965) was fitted to Kemp's ridley recapture data with a non-linear least squares regression procedure (SAS Institute, 1989) as described by Schmid and Witzell (1997). Growth models were constructed using different intervals of time between initial tagging and recapture, in order to minimize the effects of measurement errors on short-term recaptures. Furthermore, separate growth models were calculated for Kemp's ridley turtles tagged in the Gulf of Mexico and turtles tagged in the Atlantic Ocean.

Results and Discussion

Ninety-six recaptures were recorded for Kemp's ridley turtles tagged in cooperative NMFS tagging projects. Of this total, fifty-eight Kemp's ridley turtles were initially tagged in the northern Gulf of Mexico (Fla. – 31, Ms. – 1, La. – 4, and Tx. – 22), and thirty-eight were initially tagged along the Atlantic coast (Fla. – 18, Ga. – 1, N.C. – 3, Va. – 4, and N.Y. – 12). Nine Kemp's ridley recaptures were at or near Rancho Nuevo, Mexico (Table 1); four of these were initially tagged in the northern Gulf (La. – 1 and Tx. – 3) and five on the Atlantic coast (Va. – 1, N.C. – 1, and Fla. – 3). In the Gulf, a maximum recapture interval of 3.7 years was recorded for a turtle tagged in Texas and recovered in Mississippi. In the Atlantic, turtles tagged in Florida and North Carolina were observed nesting at Rancho Nuevo, Mexico 8.3 and 8.5 years after last capture.

Kemp's ridley turtle carapace lengths ranges from 22.0 to 65.9 cm for all initial captures and from 22.0 to 66.7 cm for all recaptures. A mean growth rate of 6.7 ± 6.2 cm/yr (range = 0.0 – 29.1 cm/yr) was calculated for the entire database. Carapace lengths for turtles tagged in the Gulf ranged from 22.2 to 65.8 cm for initial captures and 22.4 to 66.7 cm for recaptures. Carapace lengths for turtles tagged in the Atlantic ranged from 22.0 to 65.9 cm for initial captures and 22.0 to 66.0 cm for recaptures. The mean growth rate for Kemp's ridley turtles tagged in the Gulf (7.5 ± 6.2 cm/yr) was significantly larger (^{2} = 4.3, df = 1, p =0.04) than that of turtles tagged in the Atlantic (5.5 ± 6.2 cm/yr).

The fundamental assumption of the von Bertalanffy model is that growth rate steadily decreases with increasing size and age. Straight-line regression of individual growth rates on the mean of initial and recapture carapace lengths produced negative slopes for all records, Gulf records, and Atlantic records (Fig. 1). This suggests that Kemp's ridley turtle growth rates decreased with increasing mean carapace lengths, but correlation coefficients for the regressions were low owing to considerable variation in growth rates for 20 – 50 cm turtles.

There was very little change in the estimates of von Bertalanffy growth parameters with increasing recapture intervals, although there were increases in the standard errors of the estimates and the residual mean square errors (MSE) of the growth models (Table 2). The model for all tag records had the lowest MSE for the recapture interval data treatments and was therefore considered the most appropriate growth model (Dunham, 1978). The fitted von Bertalanffy growth equation for all recaptures combined was:

$$L_t = 70.7\ (1 - 0.9378\ e^{-0.2035\ t}) \qquad (1)$$

The estimated asymptotic length for this model is larger than the mean size (64 cm converted SSCL) and slightly smaller than the maximum size (72.5 cm converted SSCL) for nesting females observed at Rancho Nuevo, Mexico (Burchfield *et al.*, 1988).

The fitted von Bertalanffy growth equation for Kemp's ridley turtles tagged in the Gulf of Mexico was:

$$L_t = 71.1\ (1 - 0.9381\ e^{-0.2095\ t}),\ MSE = 6.6244, \qquad (2)$$

and the equation for turtles tagged along the Atlantic coast was:

$$L_t = 73.2\ (1 - 0.9399\ e^{-0.1665\ t}),\ MSE = 4.9517. \qquad (3)$$

The estimated asymptotic length for turtles captured in the Atlantic is slightly larger than that of the Gulf. This observation was probably the result of the different growth rates between the two areas and the inverse relationship of the von Bertalanffy parameters in the absence of larger individuals (Knight, 1968; Schmid and Witzell, 1997).

Estimating age to maturity has been the primary application of the von Bertalanffy growth models derived for Kemp's ridley turtles. The growth curve for all recaptures combined (equation 1) is illustrated in Figure 2. By using the mean size of nesting females (64 cm) as the size at maturity, estimates of age to maturity ranged from 11-12 years for all tagging records, 12-13 years for Atlantic records, and 10-11 years for Gulf records. Minimum size of nesting females (56.0 cm converted SSCL; Burchfield *et al.*, 1988) has also been used as an estimate of size at maturity and resulted in age to maturity estimates as early as 7-8 years for all tagging records, 8-9 years for Atlantic records, and 7-8 years for Gulf records. Another application of von Bertalanffy models has been to estimate the number of years required for turtles to grow within a specified size range (Bjorndal *et al.*, 1995; Schmid and Witzell, 1997). Accordingly, the duration of Ogren's (1989) Kemp's ridley coastal-benthic subadult stage (20 – 60 cm) would be 7 – 8 years for all tagging records and the Gulf records, and 8 – 9 years for the Atlantic records.

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Text Footnotes

[1] Bartol, S. Virginia Institute of Marine Science, P.O. Box 1346, Gloucester Point, VA 23062-1346.

[2] Branstetter, S. Gulf and South Atlantic Fisheries Development Foundation, Inc., Lincoln Center, Suite 997, 5401 West Kennedy Boulevard, Tampa, FL 33609.

[3] Corcoran, G. J.L. Scott Marine Education Center & Aquarium, P.O. Box 7000, Ocean Springs, MS 39566-7000.

[4] Epperly, S. Southeast Fisheries Science Center, Beaufort Laboratory, 101 Pivers Island Road, Beaufort, NC 28516-9722.

[5] Landry, Jr., A.M. Texas A & M University at Galveston, Department of Marine Biology, Galveston, TX 77551-5923.

[6] Martin, E. Ecological Associates, Inc., P.O. Box 405, Jensen Beach, FL 34958-0405.

[7] Nelson, D. A. U.S. Corps of Engineers, Waterways Experiment Station, 3909 Halls Ferry Road, Vicksburg, MS 39180.

[8] Rudloe, A. Gulf Specimen Marine Laboratories, Inc., P.O. Box 237, Panacea, FL 32346.

[9] Rudloe, J. Panacea Institute of Marine Science, P.O. Drawer AB, Panacea, FL 32346.

[10] Sadove, S. Puffin Consulting, Inc. P.O. Box 361, Jamesport, NY 11947; Okeanos Ocean Research Foundation, Inc., 431 E. Main Street, Riverhead, NY 11901-2556.

Table 1. Recoveries of Kemp's ridley turtles initially tagged in the U.S. and subsequently recaptured at or near Rancho Nuevo, Mexico.

Tagging Location	Date Tagged	Initial SCL	Recapture SCL	Years at Large
Atlantic				
Virginia[1]	6/19/89	46.6	64.9	6.9
Florida[2]	7/24/89	54.8	64.9	4.8
Florida[3]	1/11/89	42.8	63.8	8.3
Florida[3]	9/30/90	36.2	62.1	5.6
North Carolina[4, *]	10/26/89	40.5	63.0	8.5
Gulf of Mexico				
Texas[5]	11/15/92	33.9	35.0[7]	0.1
Texas[5]	5/1/94	56.0	65.8	3.0
Louisiana[5]	8/13/94	65.8	66.2	0.7
Texas[6,*]	3/5/98	62.2	63.0	0.2

* Not included in von Bertalanffy growth models.

[1] Bartol, S. Virginia Institute of Marine Science, P.O. Box 1346, Gloucester Point, VA 23062-1346.

[2] Martin, E. Ecological Associates, Inc., P.O. Box 405, Jensen Beach, FL 34958-0405.

[3] Schmid and Witzell (1997)

[4] Epperly, S. Southeast Fisheries Science Center, Beaufort Laboratory, 101 Pivers Island Road, Beaufort, NC 28516-9722.

[5] Landry, Jr., A.M. Texas A & M University at Galveston, Department of Marine Biology, Galveston, TX 77551-5923.

[6] Branstetter, S. Gulf and South Atlantic Fisheries Development Foundation, Inc., Lincoln Center, Suite 997, 5401 West Kennedy Boulevard, Tampa, FL 33609.

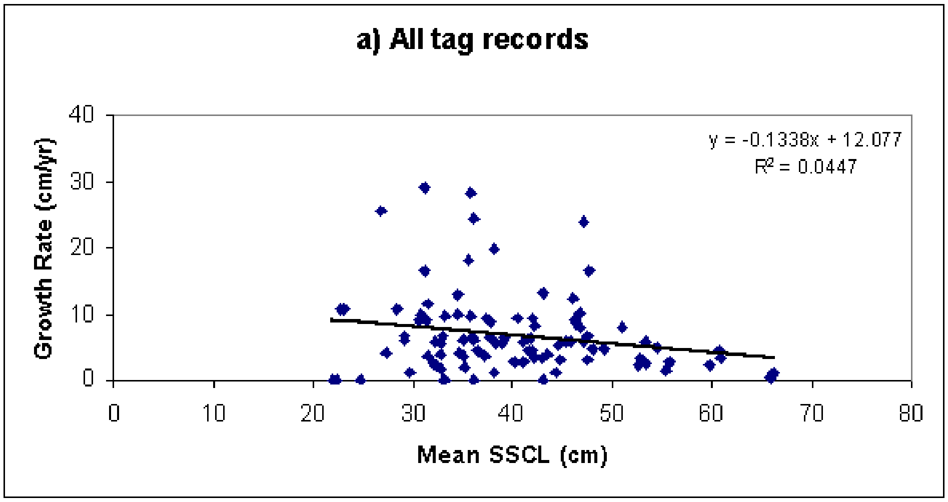
[7] Recaptured in Laguna Madre de Tamaulipas, Mexico.

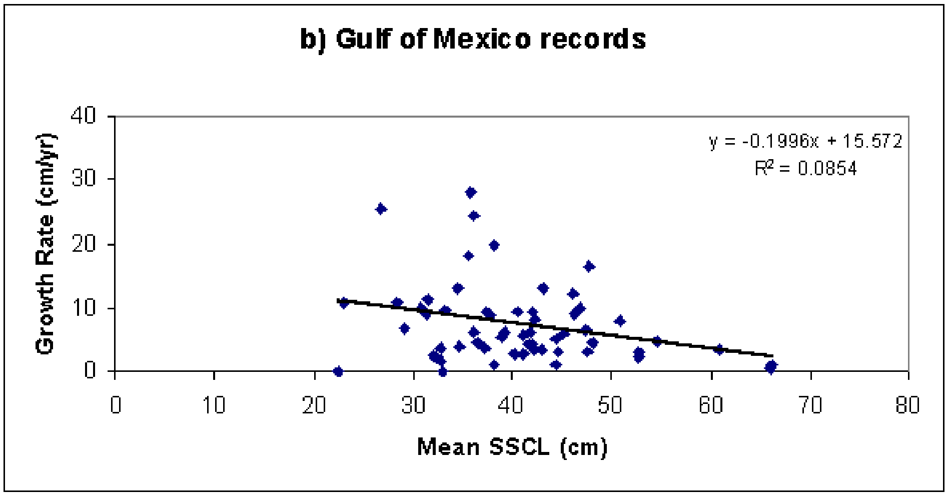
Table 2. Estimated values of asymptotic length (*a*), intrinsic growth rate (*k*), and means square error (MSE) from non-linear regression of von Bertalanffy growth interval equation for Kemp's ridley turtles (one asymptotic standard error in parentheses).

Data Treatment	*n*	*a*	*k*	MSE
All tag records	96	70.7 cm (2.7)	0.2035 (0.0251)	5.9436
All tag records > 90 days	62	70.9 cm (3.3)	0.2005 (0.0305)	8.7846
All tag records > 180 days	50	71.2 cm (3.6)	0.1953 (0.0318)	9.4246
All tag records > 365 days	20	69.7 cm (5.1)	0.2237 (0.0572)	19.5439
NMFS Miami tag records[1]	58	70.7 cm (3.4)	0.2135 (0.0325)	7.1282
NMFS Miami tag records > 90 days	38	71.0 cm (4.2)	0.2098 (0.0390)	10.3059
NMFS Miami tag records > 180 days	32	70.9 cm (4.5)	0.2103 (0.0424)	11.8087
NMFS Miami tag records > 365 days	13	69.7 cm (6.5)	0.2343 (0.0794)	26.3726

[1] Excuding the tag records used in Schmid and Witzell (1997).

Figure 1. Kemp's ridley turtle growth rates plotted by the mean of the initial and recapture carapace lengths for a) all records, b) Gulf of Mexico records, and c) Atlantic Ocean records.





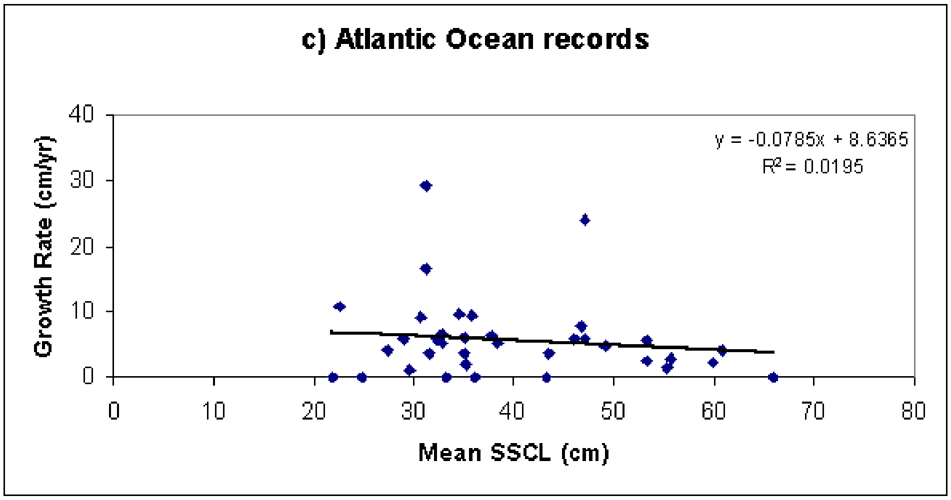
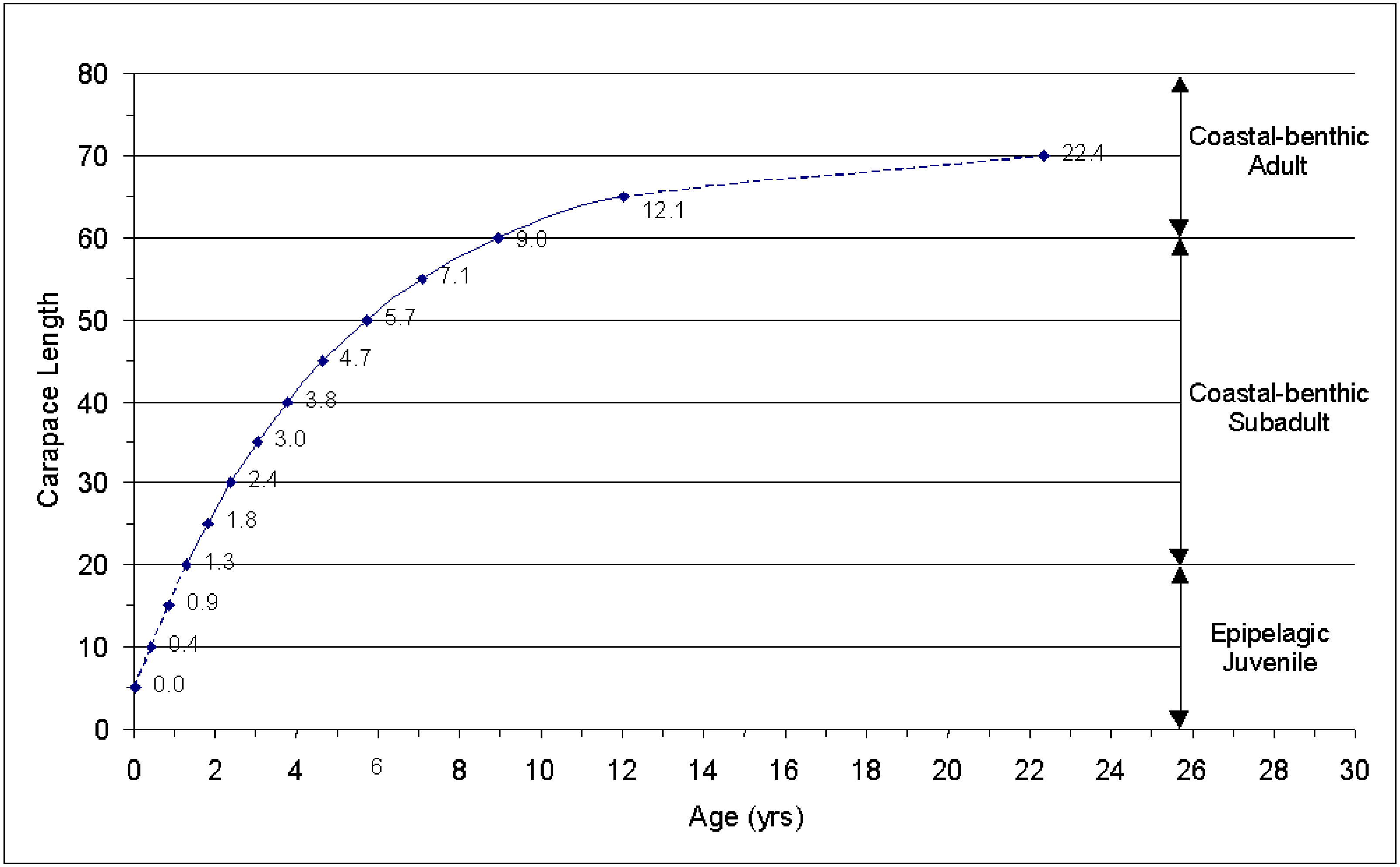


Figure 2. Growth curve for the combined database of all Kemp's ridley turtle recaptures. Dashed portion of the curve represents extrapolation outside the size range of the database. Numbers on the curve indicate the estimated age at 5 cm intervals.



Appendix 2. Sea Turtle Strandings

Figure A2.1. Statistical zones along the U.S. Atlantic and Gulf of Mexico coasts.

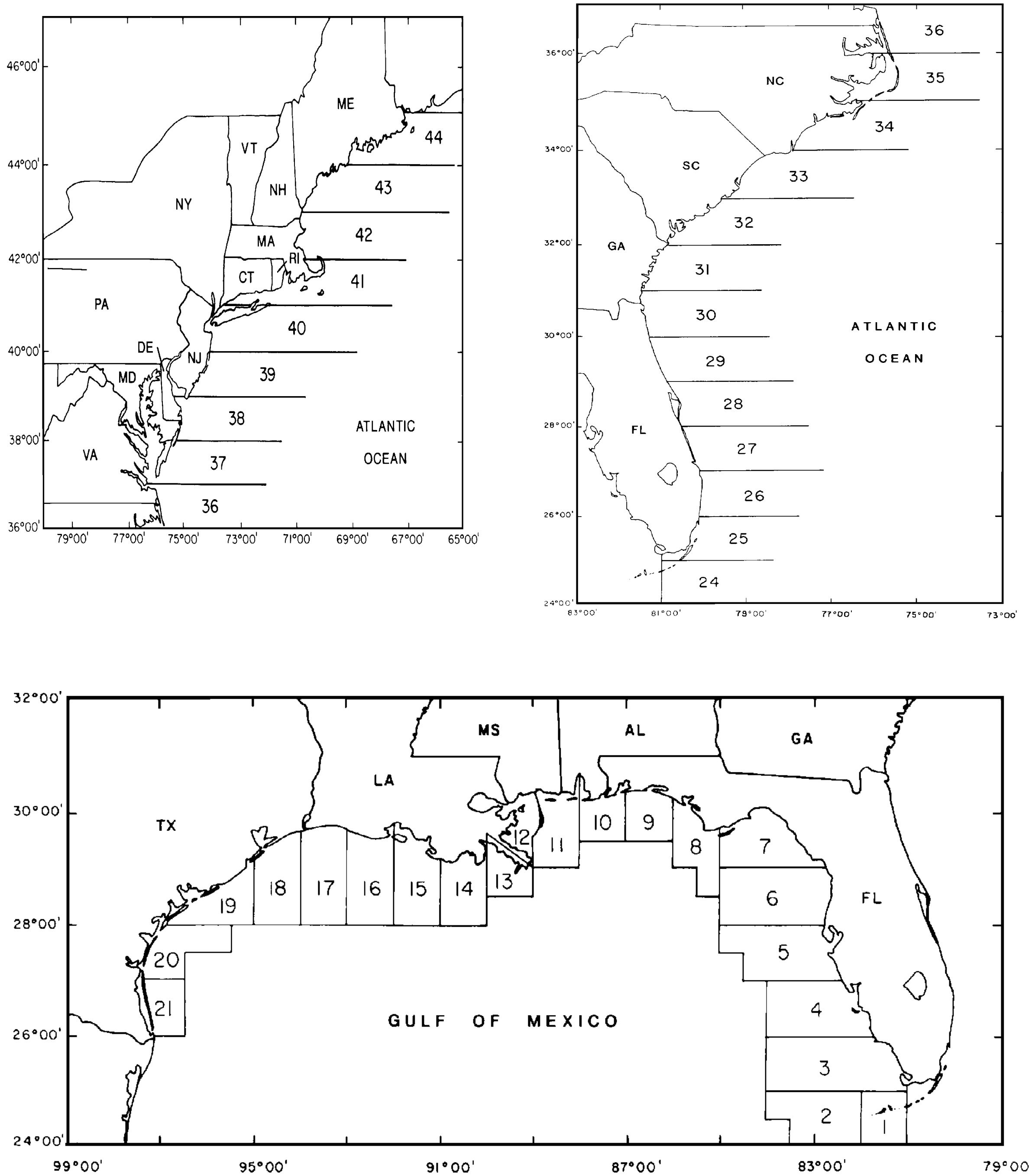


Figure A2.2a. Kemp’s ridley strandings in the U.S. Gulf of Mexico

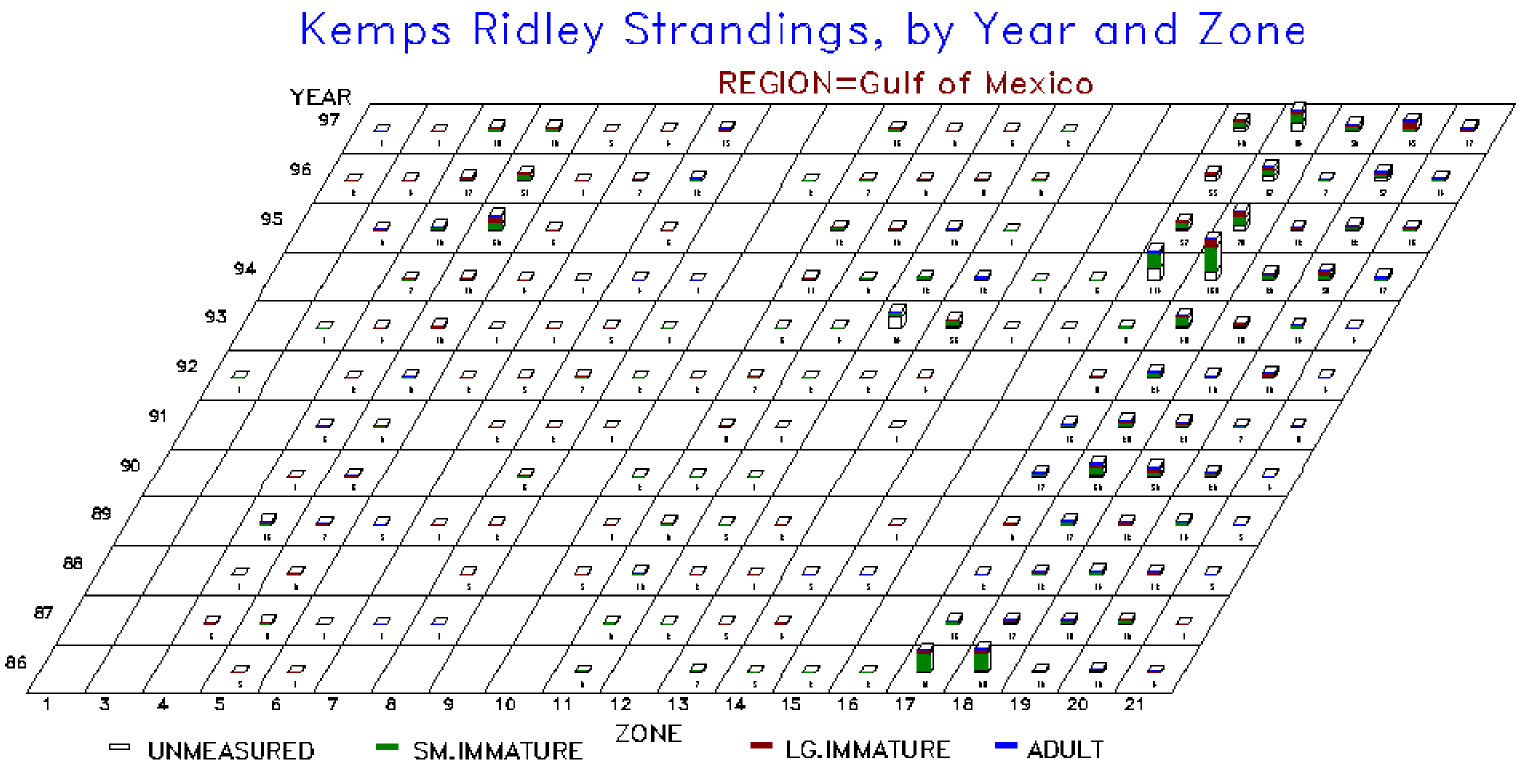

Kemps Ridley Strandings, by Year and Zone
REGION=Gulf of Mexico
YEAR
97
96
95
94
93
92
91
90
89
88
87
86
1
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
ZONE
UNMEASURED
SM.IMMATURE
LG.IMMATURE
ADULT

Figure A2.2b. Kemp's ridley strandings in the Southeast U.S Atlantic.

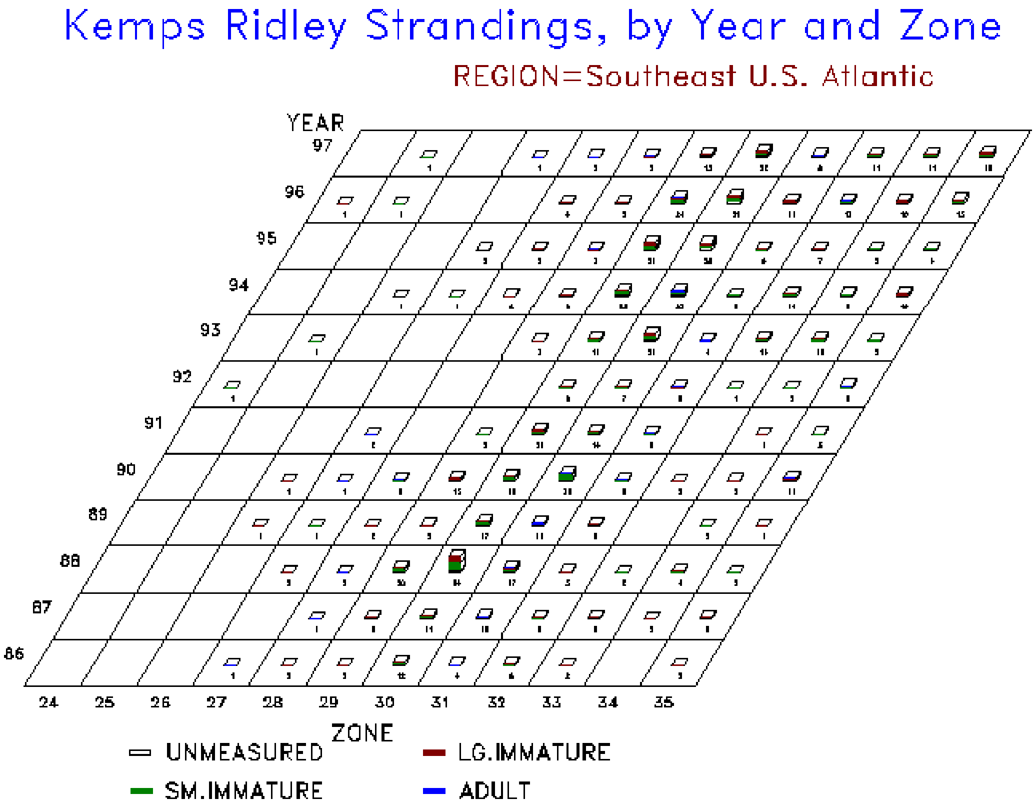

Kemps Ridley Strandings, by Year and Zone
REGION=Southeast U.S. Atlantic
YEAR
97
96
95
94
93
92
91
90
89
88
87
86
24
25
26
27
28
29
30
31
32
33
34
35
ZONE
UNMEASURED
LG.IMMATURE
SM.IMMATURE
ADULT

Figure A2.2c. Kemp's ridley strandings in the Northeast U.S Atlantic.

Kemps Ridley Strandings, by Year and Zone

REGION=Northeast U.S. Atlantic

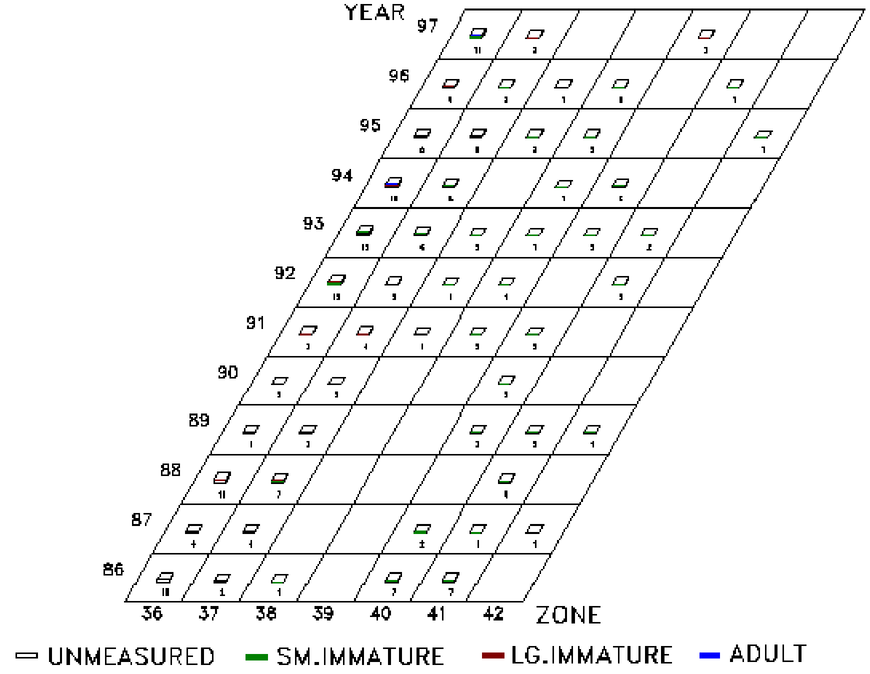


Figure A2.3a. Loggerhead strandings in the U.S Gulf of Mexico.

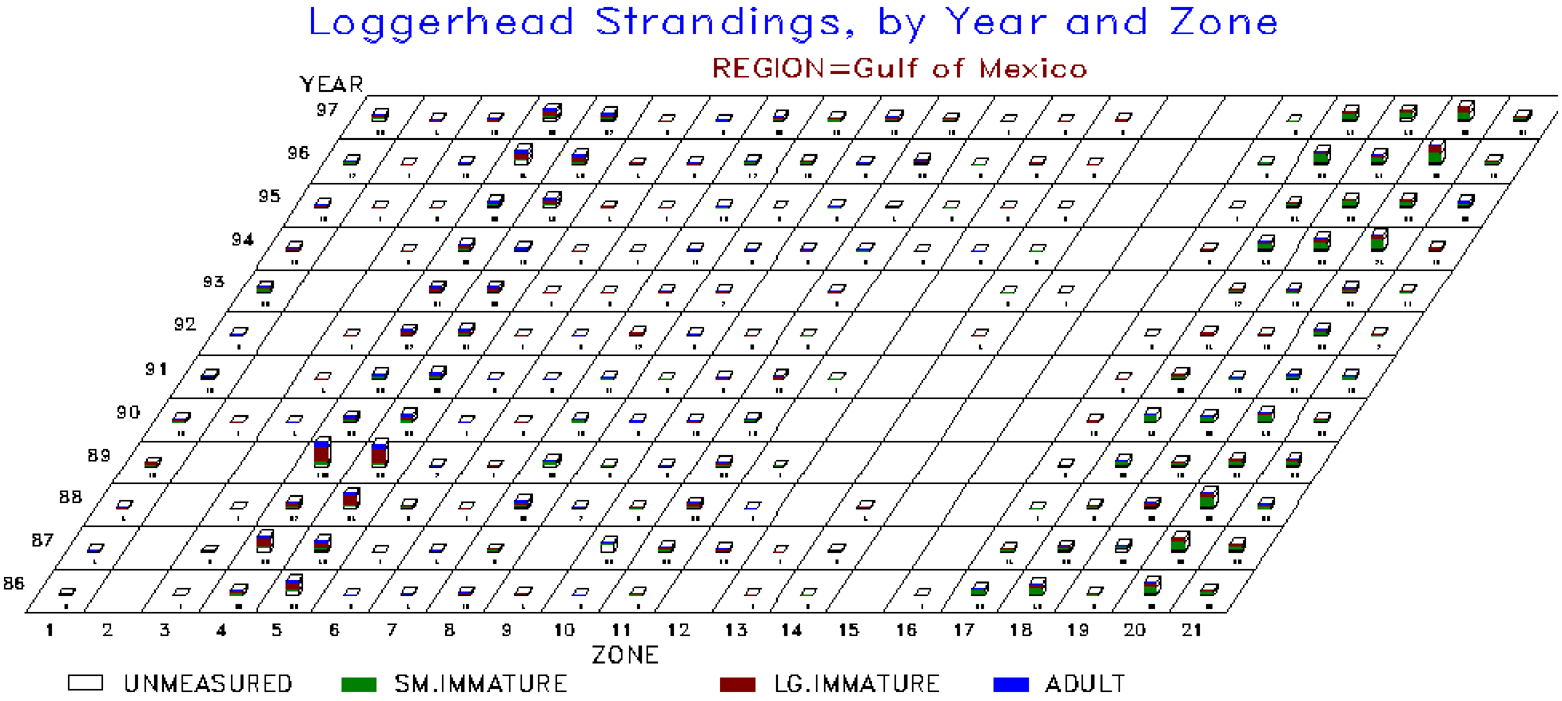

Loggerhead Strandings, by Year and Zone
REGION=Gulf of Mexico
YEAR
97
96
95
94
93
92
91
90
89
88
87
86
1
2
3
4
5
6
7
8
9
10
11
12
13
14
15
16
17
18
19
20
21
ZONE
UNMEASURED
SM.IMMATURE
LG.IMMATURE
ADULT

Figure A2.3b. Loggerhead strandings in the Southeast U.S Atlantic.

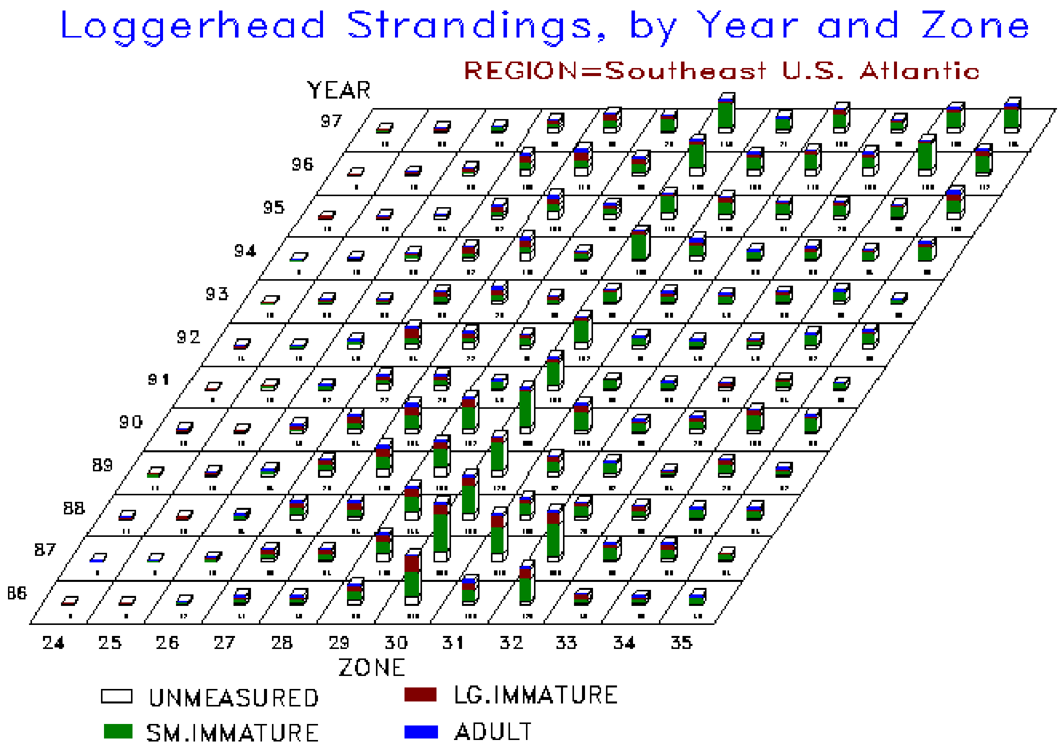
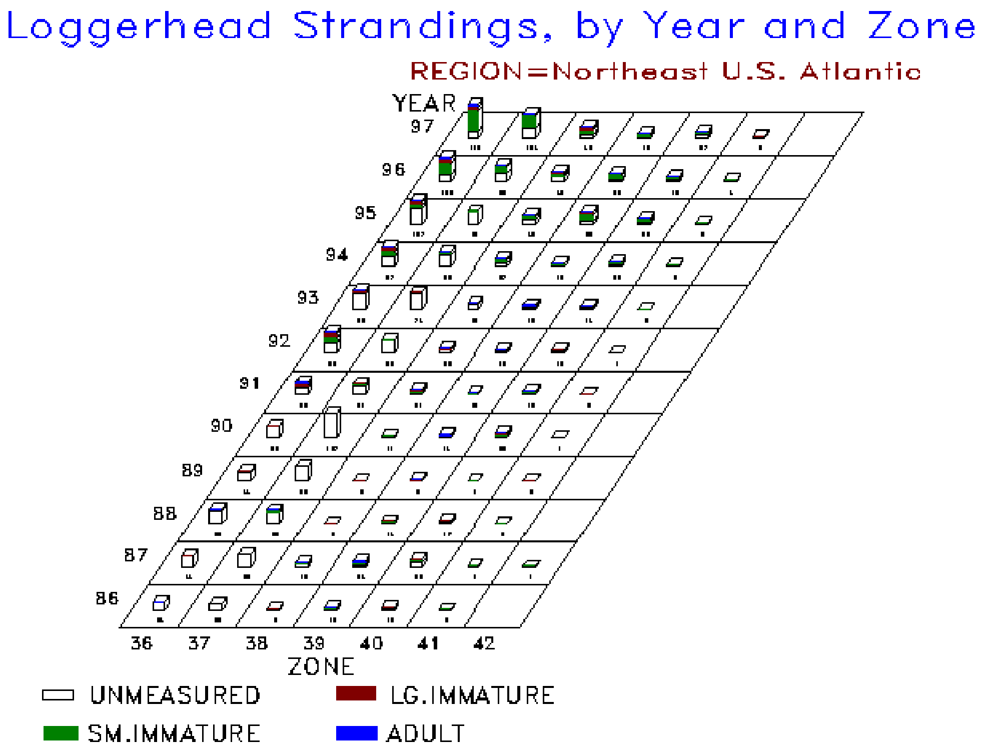

Loggerhead Strandings, by Year and Zone
REGION=Southeast U.S. Atlantic
YEAR
97
96
95
94
93
92
91
90
89
88
87
86
24
25
26
27
28
29
30
31
32
33
34
35
ZONE
UNMEASURED
LG.IMMATURE
SM.IMMATURE
ADULT

Figure A2.3c. Loggerhead strandings in the Northeast U.S Atlantic.


Loggerhead Strandings, by Year and Zone
REGION=Northeast U.S. Atlantic
YEAR
97
96
95
94
93
92
91
90
89
88
87
86
36
37
38
39
40
41
42
ZONE
UNMEASURED
LG.IMMATURE
SM.IMMATURE
ADULT