

PART I

STOCK ASSESSMENT OF LOGGERHEAD SEA TURTLES OF THE WESTERN NORTH ATLANTIC

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PART I. STOCK ASSESSMENT OF LOGGERHEAD SEA TURTLES OF THE WESTERN NORTH ATLANTIC

The Turtle Expert Working Group, established in 1995 by the Southeast Fisheries Science Center at the behest of the National Marine Fisheries Service, has published two reports (1998, 2000) which, in part, concern the status and condition of the loggerhead sea turtle stocks of the Western North Atlantic. Herein we do not attempt to duplicate material in those reports, but instead provide updated information acquired since the preparation of the last report. Thus, this document is to be used in conjunction with the two TEWG reports.

Stock Definition

Sea turtles have complex migratory behaviors and gender-specific dispersal that must be considered in defining management units. Sexual differences in dispersal or migratory behaviors may lead to different estimates of population structure calculated with mitochondrial (mtDNA) and nuclear (nDNA) DNA (Awise 1995). Bowen (1997) points out that these results are not necessarily conflicting but reflect the expected consequence of sex-specific dispersal. Assays of both biparental (nDNA) and uniparental (mtDNA) lineages are needed to understand the complex stock structure of migratory animals such as sea turtles. Either used in isolation can be misleading, especially conclusions based on nDNA alone, where in the case of sea turtles one might conclude that recruitment of females from other reproductive populations would counter the depletion of a rookery.

Assays of mtDNA illuminate the stock structure of the female lineages that are essential to reproduction and species recovery. mtDNA is used as a genetic tag to show a behavioral aspect of sea turtle life history - natal homing of egg-laying females - not to indicate important genetic differences between nesting colonies of sea turtles. Results of maternally-inherited mtDNA studies of sea turtles support the hypothesis of natal homing region (Encalada *et al.* 1996, Encalada *et al.* 1998, Bass 1999, Dutton *et al.* 1999). Each nesting assemblage represents a distinct reproductive population, regardless of the nDNA findings, because the production of progeny depends on female nesting success. Thus, should a nesting assemblage be depleted, regional dispersal will not be sufficient to replenish the depleted assemblage over ecological time scales germane to immediate management issues (Awise 1995), a consequence with both population and ecological implications. Based on mtDNA results available at the time (Bowen *et al.* 1993, Bowen 1995, Encalada *et al.* 1998), the Turtle Expert Working Group (1998, 2000) recognized at least 4 genetically distinct loggerhead (*Caretta caretta*) nesting subpopulations in the western North Atlantic and suggested that they be considered independent demographically, consistent with the definition of a distinct vertebrate population segment (59 FR 65884-65885, December 21, 1994; 61 FR 4722-4725 February 7, 1996) and of a management unit (MU) (Moritz, 1994a, b). Recent fine-scale analysis of mtDNA data from Florida rookeries indicate that population separations begin to appear between nesting beaches separated by more than 100 km of coastline that does not host nesting (Francisco *et al.* 2000¹) and tagging studies are

¹ Francisco, A.M., A.L. Bass, K.A. Bjorndal, A.B. Bolten, R. Reardon, M. Lamont, Y. Anderson, J. Foote, and B.W. Bowen. 2000. Stock structure and nesting site fidelity in Florida loggerhead turtles (*Caretta caretta*) resolved with

consistent with this result (Richardson 1982, Ehrhart 1979², LeBuff 1990, CMTTP³). Nest site relocations greater than 100 km occur, but generally are rare (CMTTP⁴, LeBuff 1974, Ehrhart 1979⁵, Bjørndal *et al.* 1983, LeBuff 1990). However, there are a number of reports of recaptured animals nesting on Little Cumberland Island, Georgia that were originally tagged in Melbourne, Florida (J. Richardson personal communication⁶). Based on these results there are at least four management units (MU) in the southeastern U.S: (1) Florida Panhandle, (2) southern Florida, (3) Amelia Island (Volusia County, Florida) and northward, and (4) the Dry Tortugas. The nesting subpopulation on the Yucatán Peninsula is a fifth MU identified in the Western North Atlantic (Encalada *et al.* 1998) and there may be more. Assemblages throughout the greater Caribbean and those in the Eastern North Atlantic (*e.g.*, Cape Verde Islands, Senegal, and Morocco; Sternberg 1981) been not been assayed, but sampling has begun in the Cape Verde Islands where a significant numbers of turtles still nest⁷.

The area between Cape Canaveral and Amelia Island has intermediate genotype frequencies that indicate another management unit by some criteria (Francisco *et al.* 1999⁸). Loggerheads nesting from Amelia Island to North Carolina are indistinguishable with mtDNA, but this means only that there is not the resolution to detect any differences, which suggests that the area was colonized by a small number of females after the last (Wisconsin) glacial epoch. Given the recent colonization northward, it is not surprising that there is insufficient genetic diversity for an assessment of stock structure. There may be different units contained in this one management unit as there are significant distances with little or no nesting between rookeries throughout the area and, based on the 100 km yardstick, likely are significantly isolated as to be

mtDNA sequences. Unpublished Manuscript . Department of Fisheries and Aquatic Sciences, University of Florida, Gainesville, 23 pp.

² Ehrhart, L.M. 1979. A survey of marine turtle nesting at the Kennedy Space Center, Cape Canaveral Air Force Station, North Brevard County, Florida. Unpublished report by the University of Central Florida, Orlando, to the Florida Department of Natural Resources, Division of Marine Resources, St. Petersburg, Fla., 122 pp.

³ Unpublished Data. The Cooperative Marine Turtle Program was established by NMFS in 1980 to centralize the tagging programs among sea turtle researchers, distribute tags, manage tagging data, and facilitate exchange of tag information. Since 1999 the CMTTP has been managed by the Archie Carr Center for Sea Turtle Research at the University of Florida, Gainesville.

⁴ *Ibid.*

⁵ Ehrhart, L.M. 1979. A survey of marine turtle nesting at the Kennedy Space Center, Cape Canaveral Air Force Station, North Brevard County, Florida. Unpublished report by the University of Central Florida, Orlando to the Florida Department of Natural Resources, Division of Marine Resources, St. Petersburg, Fla., 122 pp.

⁶ Jim Richardson, University of Georgia, Athens. Personal Communication (Phone) to Sheryan Epperly, National Marine Fisheries Service, SEFSC, Miami, Fla., January 30, 2001.

⁷ Luis Felipe López Jurado, University of Las Palmas, Cape Verde Islands, Personal Communication (E-Mail) to CTURTLE Listserv (<http://www.lists.ufl.edu/archives/cturtle.html>), January 14, 2000.

⁸ Francisco, A.M., A.L. Bass, and B.W. Bowen. 1999. Genetic characterization of loggerhead turtles (*Caretta caretta*) nesting in Volusia County. Unpublished report to Florida Department of Environmental Protection. Department of Fisheries and Aquatic Sciences, University of Florida, Gainesville, 11 pp.

MUs. We are not identifying subdivisions of the northern subpopulation as separate MUs at this time, however, there is some risk in this decision. Avise (1995) argues that a combination of genetics and demographics needs to be used to define population structure for conservation. Failure to reject the null hypothesis (panmixia) based solely on genetic parameters can lead to incorrect management decisions and managers risk losing local populations (Taylor and Dizon 1996, 1999). The identification of putative management units within the currently defined northern subpopulation as well as the entire issue of loggerhead management units is something that a recovery team needs to address immediately.

Nuclear DNA contains the important genes for adaptation and long-term survival. Since it is biparentally inherited it provides information on the behavior of male sea turtles that is not available from mtDNA. In 1999 NMFS contracted the Department of Fisheries and Aquatic Sciences, University of Florida, to analyze nDNA data from loggerhead rookeries in the western Atlantic. A final report is due soon and the results will be presented at the upcoming meeting of the American Society of Ichthyologists and Herpetologists in July 2001. Very preliminary results indicate that population structuring defined by nDNA (microsatellite) assays is much lower in the southeast U.S. than found in the mtDNA studies⁹. The implication is that males are a conduit for gene flow between the egg-laying populations defined by female site fidelity, but the amount of male-mediated gene flow is not yet determined. Three points need to be made: (1) The population structuring observed with nDNA, while lower than observed with mtDNA, may still be significant across the southeast U.S., supporting the subdivision into multiple stocks, (2) A little male-mediated gene flow between nesting colonies means that concerns about genetic diversity within nesting populations may be less pressing and small nesting populations are less likely to suffer the effects of inbreeding, and (3) These conclusions about nDNA of western North Atlantic loggerhead sea turtles are extremely preliminary and further analysis of the data is ongoing. The results of a study on loggerheads in the eastern Mediterranean demonstrated there was low male-mediated gene flow between nesting sites and that there was genetic substructuring due to the high precision of natal homing by nesting females (Schroth *et al.* 1996). These authors concluded that in order to preserve the genetic diversity of the *Caretta* metapopulation in the eastern Mediterranean one needed to preserve individual nesting sites.

Foraging grounds contain cohorts from nesting colonies from throughout the Western North Atlantic (see Table 10 in TEWG 2000). Since the preparation of the last TEWG report, three more reports have provided additional genetic data on the foraging ground composition of loggerhead sea turtles. The Florida Bay loggerhead foraging population is composed primarily of individuals from the South Florida subpopulation (84%) with some contribution observed from the northern subpopulation (8%), the Florida Panhandle subpopulation (<1%), and the Yucatán subpopulation (8%) (Bass *et al.* 1998¹⁰). Additional samples from North Carolina's

⁹ Discussions (E-Mail) between Brian Bowen (contractor, University of Florida, Gainesville) and Sheryan Epperly (contract technical monitor, National Marine Fisheries Services, SEFSC, Miami, Fla.) and analyses by Alicia Francisco (graduate student of Dr. Bowen), November 7, 2000, November 14, 2000, December 17, 2000, and December 29, 2000.

¹⁰ Bass, A.L., M. Clinton, and B.W. Bowen. 1998. Loggerhead turtles (*Caretta caretta*) in Florida Bay: an assessment of origin based on genetic markers. Unpublished report to Florida Department of Environmental Protection. Department of Fisheries and Aquatic Sciences, University of Florida, Gainesville, Fla., 5 pp.

Pamlico-Albemarle Estuarine Complex revealed that the South Florida subpopulation dominated (64%) there (Bass *et al.* 2000¹¹). The northern subpopulation contributed 30% and the remaining were divided among Mexico (5%) and possibly Brazil (<1%). The authors also found significant temporal variation in the relative contributions of the subpopulations among the 3 yrs.

In 1998 NMFS contracted the analysis of samples collected from stranded animals throughout the southeastern U.S. and Gulf of Mexico¹². Those results (Bass *et al.* 1999¹³) (Fig. 1) indicate that the relatively large South Florida subpopulation dominates everywhere, but more so in the Gulf of Mexico where in the western Gulf it accounts for 83% of the animals. In Florida (geographic distribution of sampling unknown), the contribution of the South Florida subpopulation was 73%. In Georgia, its contribution was 73%, also (*Ibid.*). The contribution of this subpopulation decreased to 65-66% off the Carolinas and decreases further north of Cape Hatteras (46%). In the northernmost area sampled, Virginia, the northern subpopulation accounted for 46% of the animals. It contributes 25-28% off the Carolinas, 24% off Georgia, and off Florida east and west coast combined, contributes 20%. The contribution of the northern subpopulation to western Gulf cohorts is but 10%. The Yucatán subpopulation's contribution throughout the region ranged from 6-9%, except off Georgia where the contribution was but 3%. The Florida Panhandle subpopulation was not included as a possible contributor in these analyses because it is unlikely that its contribution could be detected against the hundreds of individuals assayed from South Florida; the inclusion of populations that contribute less than 1% in the overall nesting effort generates overestimates of contribution and can compromise the accuracy of estimates made for the other source populations.

Other sources of information indicate structuring of the Western Atlantic nesting assemblages of loggerhead sea turtles. Results of a study on carapace epibionts on turtles nesting along the Atlantic Coast of the U.S. indicated there were two populations of turtles, divided at northeast Florida (Cape Canaveral to Daytona Beach) (Caine 1986). The epibiont community included a number of long-lived sessile organisms likely unaffected by short term immigration or emigration. The low amount of overlap in the epibiont communities (4.2-7.5%) indicated that turtles were spending time in different foraging environments. Certain epibionts of the southern population of nesting turtles were of Caribbean origin whereas some of the epibionts of the northern nesting turtles were indicative of the Sargasso Sea. Based on recent satellite telemetry studies and on returns of tags, both applied at nesting beaches, non-nesting adult females from the South Florida subpopulation are distributed throughout the Bahamas, Greater Antilles, Cuba, Yucatán, eastern Gulf of Mexico, and southern Florida (Meylan 1982, Meylan *et al.* 1983,

¹¹ Bass, A.L., S.P. Epperly, J. Braun-McNeill, and A. Francisco. 2000. Temporal variation in the composition of a loggerhead turtle (*Caretta caretta*) developmental habitat. Unpublished manuscript. Department of Fisheries and Aquatic Sciences, University of Florida, Gainesville, Fla., 26 pp.

¹² The Sea Turtle Stranding and Salvage Network is a cooperative endeavor between NMFS, other federal agencies, the states, many academic and private entities, and innumerable volunteers. Data are archived at the National Marine Fisheries Service Southeast Fisheries Science Center in Miami, Fla.

¹³ Bass, A.L., S-M. Chow, and B.W. Bowen. 1999. Final report for project titled: genetic identities of loggerhead turtles stranded in the Southeast United States. Unpublished report to National Marine Fisheries Service, order number 40AANF809090. Department of Fisheries and Aquatic Sciences, University of Florida, Gainesville, Fla., 11 pp.

<http://cccturtle.org/sat1.htm>) whereas non-nesting adult females from the northern subpopulation appear to occur almost exclusively along the east coast of the U.S. (<http://cccturtle.org/sat1.htm>); only one northern subpopulation mature female has been reported to enter the eastern Gulf of Mexico (Bell and Richardson 1978), and none have been reported from international waters (CMTTP³). Limited tagging data suggest those adult females nesting in the Gulf of Mexico that are not part of the South Florida subpopulation remain in the Gulf of Mexico, including on feeding grounds off Yucatán (Meylan 1982, <http://cccturtle.org/sat1.htm>). Annual nesting at rookeries within a subpopulation's nesting range is correlated, but nesting among subpopulations is not (TEWG 2000).

Status and Trends

Nesting beaches

The preparation of the TEWG reports (1998, 2000) pre-dated the identification of the Dry Tortugas as a management unit. The reader is referred to the TEWG reports for discussions on the other subpopulations.

Dry Tortugas

Sea turtle nesting in Dry Tortugas National Park is the highest in all of Monroe County, which encompasses all of the Florida Keys (Reardon 2000¹⁴) (Fig. 2). The second highest productive nesting area in the Florida Keys is the Marquesas Keys (Florida Fish and Wildlife Conservation Commission 2000¹⁵), 47 miles east of the Dry Tortugas. The genetic affinity for individuals in the Marquesas Keys as well as the rest of the Florida Keys has yet to be assayed. The Dry Tortugas is a group of seven islands with accompanying marine habitats, 70 miles (113 km) west of Key West, Florida. Since 1995 the beaches of all 7 islands were patrolled daily from early April through late October. The full extent and status of the Dry Tortugas subpopulation is unknown at this time. Two of the seven islands, East Key and Loggerhead Key, are host to 90% of all nesting activity observed in the Park (Reardon 2000¹⁴). In the early 1980's a tagging study was conducted on the nesting turtles of East Island and the nesting population was estimated at 40 individuals (Dawson 1985¹⁶). Nesters ranged from 78.5 to 99.0 cm straight carapace length with a mean length of 90.4 cm³ (Fig. 3). The range in annual number of recorded nests for the period 1995-2000 was 190-269 with a mean of 217 nests/year (Table 1). The average clutch size has ranged from 98-105 eggs annually with an incubation time ranging from 51.0 to 54.6 days

¹⁴ Reardon, R.T. 2000. Annual Report - 2000 Season. Dry Tortugas National Park Sea Turtle Monitoring Program, Monroe County, Florida. Unpublished report. Annual report by Florida International University to Dry Tortugas National Park, Miami, Fla., 49 pp.

¹⁵ Florida Fish and Wildlife Conservation Commission. 2000. Statewide Nesting Beach Survey Program Database. Reported Nesting Activity of the Loggerhead Turtle, *Caretta caretta*, in Florida, 1993-1999. Unpublished Report. Florida Marine Research Institute, St. Petersburg, Fla., 8 May 2000, 26 pp.

¹⁶ Dawson, R.H. 1985. Project completion report: results of the 1985 sea turtle nesting survey at Fort Jefferson National Monument, Dry Tortugas, Florida. Prepared by the National Park Service Southeast Regional Office, Atlanta, Georgia for USFWS Endangered Species Field Station, Jacksonville, Fla., 49 pp.

(average 52.9 days). Eighty percent of loggerhead clutches were inventoried and hatching success ranged from 72.3 to 82% annually with an average of 77.1%.

Due to the relative isolation and lack of fresh water these islands are without mammalian nest predators but about 10% of the nests are lost annually to erosion¹⁴. Local potential threats to nesting in the Park is mainly limited to visitation; human usage needs to be monitored particularly during the nesting season to limit impacts to nests (Reardon 2000¹⁴, Dawson 1985¹⁶). Within the Park commercial fishing is prohibited and recreational fishing is limited. Presently the surrounding marine habitats are being considered for a designation as an Ecological Reserve. Although the proposed Tortugas Ecological Reserve concentrates on protecting the coral reef ecosystem and fish stocks, resident turtles and migratory nesters should benefit also due to the intended expansion of “no take” zones (U.S. Department of Commerce¹⁷).

Nesting Trends

Previous estimates of nesting trends for the northern subpopulation prior to the implementation of TEDs are a decline of 3 percent per year ($\lambda=0.97$) (Frazer 1983b) for Little Cumberland Island, Georgia and a decline of 5% per year ($\lambda=0.95$) (TEWG 1998) for South Carolina. It is possible that these two beaches are not representative of the overall subpopulation trend as Little Cumberland Island is known to be a highly erosional beach and nesting at Cape Island, the largest rookery in South Carolina (and in the northern subpopulation), may have been affected by raccoon predation control in the first half of the 20th century (S. Murphy personal communication¹⁸). For the south Florida population, Hutchinson Island, Florida was increasing at 2.2 percent per year prior to the implementation of TEDs (TEWG 1998).

Regression analysis of individual beaches in the northern subpopulation revealed both significantly positive and negative trends on some of the beaches. To assess these trends simultaneously, nesting data from selected beaches were used in a meta-analysis to estimate changes in nesting activity over time for the northern subpopulation and the South Florida subpopulation (Appendix 1). The data were limited to sites where surveys were believed to have been relatively constant over time. It is an unweighted analysis and does not consider the beaches' relative contribution to the total nesting activity of the subpopulation and must be interpreted with some caution. The analysis treats nesting beaches as random samples from the total. It is necessary to have information on relative abundance in each nesting site in order to obtain an unbiased overall trend for the populations as a whole.

The pre-1990 northern subpopulation growth rate calculated in the meta-analysis varied, depending on the statistical assumptions one makes, from not significantly different from $r = 0.0$ or $\lambda = 1.0$ ($r=\ln(\lambda)$ ($r = -0.026$, $SE = 0.105$) to a value ($r = -0.030$, $SE = 0.012$) similar to the rate reported previously for Little Cumberland Island. After 1990, the analysis indicates an

¹⁷ U.S. Department of Commerce. 2000. Strategy for stewardship: Tortugas Ecological Reserve. Final Supplemental Impact Statement/Final Supplemental Management Plan. National Oceanic and Atmospheric Administration, Washington, DC., 310 pp.

¹⁸ Sally Murphy, South Carolina Marine Resources Department, Charleston, S.C. Personal Communication (E-Mail) to Sheryan Epperly, National Marine Fisheries Service, SEFSC, Miami, Fla., October 4, 2000.

increasing trend of 2.8-2.9% per year. These results should represent the best-case scenario as the effect of Cape Island nesting activity is dampened in the unweighted analysis.

For the south Florida subpopulation, $r = 0.054-0.055$ (SE = 0.022, 0.014), and it was increasing at 5.3-5.4% per year 1979-1989. Although the subpopulation has been increasing since 1979, the meta-analysis of nesting trends indicates a slowing in the rate of that increase to 3.9-4.2% per year after 1989, but this is not significantly different from the pre-1990 rate. An important caveat for population trends analysis based on nesting beach data is that this may reflect trends in adult nesting females, but it may not reflect overall population growth rates well. Adult nesting females often account for less than 1% of total population numbers.

In-water Surveys

Fishery independent, in-water studies of sea turtles have been carried out at multiple sites in the Gulf and U.S. Atlantic with varying goals and target species. To date, NMFS has not been able to use the results of these studies to determine trends of in-water sea turtle populations (see discussion in TEWG 2000). In March 2000 NMFS sponsored a workshop to determine the feasibility of using sea turtle catch and survey methods to determine relative population abundance and population trends and to train participants in analyzing their data for this purpose (Bjorndal and Bolten 2000). The participants concluded that although the duration to detect trends in relative abundance differed among studies, all techniques reviewed appeared to be feasible. However, many have not been standardized over a long enough period to analyze for trends. Furthermore the statistical power varied among the studies. A fishery-dependent trawl survey examined was an exception – it did not appear to be a feasible method - but those data were not examined with non-parametric statistics. Epperly (in Bjorndal and Bolten 2000) clearly demonstrated the value of using non-parametric statistical models in the analysis of data sets with a large number of zero catches, which is typical of random sampling for sea turtles. As sufficient data are accumulated we encourage researchers to begin publishing the results of their studies to elucidate trends in abundance of non-nesting turtles.

Trends: Southeast Area Monitoring and Assessment Program - (SEAMAP)

In 1986, the South Carolina Marine Resources Department initiated a NMFS-funded fishery-independent trawl survey off the southeastern U.S. states to assess finfish populations (SCMRD 2000). The survey includes ocean waters 15-60 ft (4.6-18.3 m) deep, from Cape Hatteras to Cape Canaveral (Fig. 4). In 1990 the survey was standardized and stations were chosen based on a stratified random design and once established were fixed and trawled repetitively over the years. The survey design is to make 78 tows/season in nearshore strata in spring, summer, and fall and 27 and 16 tows in offshore strata in spring and fall, respectively. Over the 11 yr period of 1990-2000, only 10 stations have been missed. Paired 75 ft (22.9 m) high rise trawls (Mongoose-Falcon nets), originally of 1-7/8 in (4.8 cm) stretch mesh and in later years of 1-5/8 in (4.1 cm) mesh, without turtle excluder devices have been used throughout the study, and with very few exceptions tow duration for each haul has been 20 min. during daylight hours. Sea turtles infrequently are captured. The survey now is more than a decade old and is analyzed for trends in loggerhead sea turtle abundance for the first time.

Methods

The process of calculating the indices of abundance from this data involves the standardization of yearly changes in bycatch rate, accounting for the influence of those factors that have a significant influence. Factors which were considered as possible influences on bycatch rates included year, season, latitude, and precipitation state during the tow (PRECIP, rated as none, light rain, or moderate rain), surface salinity, bottom salinity, air temperature, surface water temperature, bottom water temperature, barometric pressure, time at the start of the tow, water depth at the start of the tow, and vectors of wind velocity from the north (NORTWIND, typically along shore) and from the west (WESTWIND, typically off shore). Effort units were defined as the individual tows, which as stated previously were nearly all 20 minutes in duration.

The areas defined for the survey are shown in Figure 4. Area strata were categorized as either INNER (nearer to shore) or OUTER (further from shore). Few turtles were caught in the OUTER strata; when this did occur, it was usually during the spring season. Preliminary examination of the data suggested that this OUTER turtle bycatch during the spring might result from colder temperatures in the INNER strata during that year-season, with turtles consequently staying in the deeper waters. Furthermore, turtle migration takes place during the spring season, which may result in bycatch levels which are subject to local migration patterns rather than reflective of abundance. For these reasons, the analysis data set was restricted to the INNER area strata and to the summer and fall seasons. The observed loggerhead turtle yearly bycatch rates are shown in Table 2 and Figure 5.

The Lo method (Lo *et al.* 1992) was used to develop standardized indices; with that method separate analyses are conducted of the positive bycatch rates and the proportions of the observed tows on which turtles were caught. This has been used previously for analyses of bluefin tuna catch rates on rod and reel (Ortiz *et al.* 1999, Turner *et al.* 1999, Brown *et al.* 1999), catch rates which are similar to the turtle bycatch rates from the SEAMAP survey data in that they can be extremely low, particularly for the largest size classes of bluefin tuna. For those bluefin tuna analyses, a delta-lognormal model approach was used; this used a delta distribution with an assumed binomial error distribution for the proportion of positive observations (trips), and assumed a lognormal error distribution for the catch rates on successful trips. More recent analyses for bluefin tuna rod and reel catch rates (Brown in prep) and yellowfin tuna longline catch rates (González Ania *et al.* 2001) used a delta-Poisson model approach, differing from the delta-lognormal approach in that a Poisson error distribution is assumed for the catches on successful trips. The delta-Poisson model approach was used for the analyses of the turtle bycatch rates.

Parameterization of the model was accomplished using a Generalized Linear Model (GLM) structure: The proportion of tows with loggerhead bycatch (*i.e.*, positive observations) per stratum was assumed to follow a binomial distribution where the estimated probability was a linearized function of fixed factors. The logit function linked the linear component and the assumed binomial distribution. Similarly, the estimated catch observed on positive trips was a function of similar fixed factors with the log function as a link.

A stepwise approach was used to quantify the relative importance of the main factors explaining the variance in bycatch rates. That is, first the Null model was run, in which no factors were entered in the model. These results reflect the distribution of the nominal data. Each potential factor was then tested one at a time. For each run, the deviance was calculated as the negative of twice the difference between the log-likelihood under the model and the log-likelihood under the maximum achievable (saturated) model:

$$D^*(y; \hat{u}) = -2(l(\hat{u}; y) - l(\hat{u}_{\max}; y))$$

The results were then ranked from greatest to least reduction in deviance per degree of freedom when compared to the Null model. The factor which resulted in the greatest reduction in deviance per degree of freedom was then incorporated into the model, provided two conditions were met: (1) the effect of the factor was determined to be significant at at least the 5% level based upon a χ^2 (Chi-Square) test, and (2) the deviance per degree of freedom was reduced by at least 1% from the less complex model. This process was repeated, adding factors (including factor interactions) one at a time at each step, until no factor met the criteria for incorporation into the final model. The final model then, included any significant fixed and random (year)*factors interactions.

The product of the standardized proportion positives and the standardized positive catch rates was used to calculate overall standardized catch rates. For comparative purposes, each relative index of abundance was obtained dividing the standardized catch rates by the mean value in each series.

Results and Discussion

The results of the stepwise procedure to develop the models are shown in Table 3 for the proportion positive bycatch model and in Table 4 for the positive bycatch model. The factors examined did not explain much of the catch rate variability in either model. For the proportion positive bycatch model, only the factor of latitude (LAT) met the conditions required for inclusion in the model (significance at the 5% level and reducing deviance per degree of freedom by at least 1%). The factor YEAR was included in the final model since this was the factor of concern and for which the least-square means were to be calculated. Together, LAT and YEAR accounted for only a 4.4% reduction in deviance per degree of freedom from the NULL model. For the positive bycatch model, none of the tested factors met the conditions required for inclusion in the model. This is not surprising, since there is very little contrast in the positive catch data; nearly 95% of the positive catch observations were of 1 turtle, with remainder being 2 turtles caught per tow. Again, YEAR was included in the final model in order to calculate the least square means. Although the positive catch analysis results are unreliable due to the lack of contrast, the end result is that values close to the nominal positive catch rates are combined with the results of the proportion positive analysis to produce annual index values. Therefore, the conclusions are primarily based upon the proportion positive analysis.

The results of the model fits for the updated indices are shown in Table 5 for the proportion positive bycatch model and in Table 6 for the positive bycatch model. The index values are shown in Table 7 and in Figure 6. The relative observed bycatch rates are also shown in Figure 6. It is clear that the standardized trend varies little from the nominal trend. However, the standardization procedure does provide some measure of the uncertainty around the relative

indices calculated from this survey. This permits the calculation of the power of this survey to detect changes in abundance.

It does appear that the catches have been increasing; a regression analysis indicated an increasing trend of 11.2%/yr relative to the catch during the first year. However, the error about each year's point estimate is large and the number of captures in 2000 is not significantly different than the number captured in 1990 ($p=0.24$). Thus, no significant trend was detected in this fishery-independent survey to indicate that the in-water population of loggerheads in the Western North Atlantic is increasing.

We assessed the power of the SEAMAP monitoring program to detect a trend in loggerhead sea turtle abundance by utilizing the program TRENDS. At a recent workshop on in-water sea turtle population trends held in March 2000 (Bjorndal and Bolten 2000) the emphasis was on minimizing the Type II error (maximizing power to detect trends) so the Type I error was set to 0.2 and the Type II error to 0.1. For purpose of comparison to the results of that workshop, we used the same criteria and ran two trials. Trial A was to determine the minimum detectable annual rate of change within the 11 years duration of this program, assuming population growth is exponential and declining, Type I Error (α)=0.2, and Type II Error (β)=0.1, the statistical power = 0.9. Trial B was to determine the minimum duration (yrs) required to detect an annual decline of 25%. These analyses indicated that the SEAMAP monitoring program could detect a trend of -0.24%/year after 11 yrs, the same amount of time required to detect a decline of 25%/year. Therefore, unless the population was changing in size at about 25% per year, it is unlikely (<90% probability) that the SEAMAP monitoring program would be able to detect a trend within the duration that it has been ongoing (11 yrs).

Stock Assessment

Crouse *et al.* (1987) developed the first stage-based matrix population model for the loggerhead turtle. They collapsed Frazer's (1983a) 54-stage loggerhead life-table into 7 stages, hatchlings, small juveniles, large juveniles, subadults, novice breeders, 1st year remigrants and adults. In a further refinement of the model, Crowder *et al.* (1994) reduced the 7-stage model to a 5-stage model, combining all breeding adults into one stage. Crowder *et al.* (1994) also presented an age-based matrix model of loggerheads in order to qualitatively assess how population trajectories respond to management practices.

Heppell *et al.* (in press) redefined the stages first changing the model from a post-breeding census to a pre-breeding census, incorporating first year survival into the fertility term and eliminating hatchlings as a separate stage. In addition, Heppell *et al.* (in press) eliminated the subadult stage and defined three juvenile stages, pelagic juveniles, small benthic juveniles and large benthic juveniles. TEWG (1998) defined the cutoff between small and large benthic juveniles at 70 cm straight carapace length (SCL) based on differential habitat utilization. Loggerheads slightly larger than 70 cm may be too large to fit through the smallest current TED

openings¹⁹, introducing potentially different mortality rates between the two benthic juvenile stages. Because current regulations require smaller TED openings in the Gulf of Mexico than in the Atlantic, this cutoff can be a bit fuzzy, but large juveniles and adults probably experience limited benefits from TEDs. Heppell *et al.* (in press) used 70 cm SCL as the cutoff between small and large benthic juveniles. Another change from the previous models is that a variable remigration interval is incorporated, making nesting females a separate stage from non-nesting females. As in Crowder *et al.* (1994), Heppell *et al.* (in press) expanded the model to be age-based in order to assess population responses to TED regulations. The model, then, is essentially a Leslie matrix, with annual survival rates on the subdiagonal and fecundity in the top row. The row of the matrix equivalent to age at reproductive maturity represents breeding females. The remaining 4 rows of the matrix cycles the surviving neophytes and remigrants based on the proportion of females returning to nest after 1, 2, 3, 4, or 5 years which are 3%, 56%, 31%, 7% and 3% (Richardson *et al.* 1978).

The models we present here are the same as the 5-stage structured models of Heppell *et al.* (in press) and are similarly expanded to age-based models. However, to update the parameters of the models as much as possible we analyzed new data sets to determine the best available information to use in this current stock assessment. We construct models using both the historical and updated vital rates.

Vital Rates

Duration of Stages

Heppell *et al.* (in press) present two models, both incorporating the structure described above. Model 1 uses stage durations that are consistent with the previous models and derive from a von Bertalanffy growth curve developed by Frazer (1987). Model 2 uses longer stage durations that are based on a von Bertalanffy growth curve developed from a preliminary analysis of a mark-recapture study in North Carolina (Braun-McNeill *et al.* in press). Since Frazer's (1987) growth model was based on loggerheads caught in Florida, we thought that Model 1 might be representative of a faster growing population in the south, and model 2 representative of a slower growing northern population (see previous section on stock definition).

To further assess individual growth rates and the possibility of regional variability, we analyzed published von Bertalanffy growth curves that were based on mark-recapture data from wild loggerheads in the southeast U.S. (Table 9, Fig. 7). The curves prepared by Braun-McNeill *et al.* (in prep)²⁰ used data for turtles whose time between first capture and recapture was greater than 11 months. Schmid (1995) prepared a curve where he only used recaptures when the time

¹⁹ Epperly, S.P and W.G. Teas. 1999. Evaluation of TED opening dimensions relative to size of turtles stranding in the Western North Atlantic. U.S. Department of Commerce, National Marine Fisheries Service SEFSC Contribution PRD-98/99-08, Miami, Fla, 31 pp.

²⁰ Braun-McNeill, J., S.P Epperly, and L. Avens. A preliminary analysis of growth rates of immature loggerhead (*Caretta caretta*) sea turtles from North Carolina, U.S.A. Manuscript in preparation.

between first capture and recapture was greater than or equal to 365 days (designated as ** in Table 8). As with other poikilotherms, metabolic activity in loggerheads is external-temperature dependent and it is likely that little or no growth occurs during the winter months (Castanet 1994), hence inferring growth rates from time periods of less than one year may result in inaccuracies. The Schmid (1995)** curve was prepared from only 19 growth rates and it is unclear what the size distribution was for turtles included in the analysis. The Braun-McNeill *et al.* (in prep)²⁰ curve was estimated primarily from small benthic juveniles, using growth rates from 57 turtles. In order to apply the Braun-McNeill *et al.* (in prep)²⁰ curve to the entire benthic life-stage, we extended the size range by adding additional mark-recapture growth rates for animals greater than 70 cm SCL from the CMTTP²¹. We used records from both data sets for animals that were at large for at least 0.9 yr, had a straight carapace length recorded, and did not indicate negative growth. From the CMTTP, in order not to bias the growth curve to the growth rates of a few individuals, we used only one growth rate for each animal included, even if there were multiple recapture records for the animal (Fig. 8 and 9).

Chaloupka and Limpus (1997) and Limpus and Chaloupka (1997) found sex-specific growth rates in hawksbill (*Eretmochelys imbricata*) and green (*Chelonia mydas*) sea turtles in the southern Great Barrier Reef. The turtles in these studies were sexed by internal observation of the gonads. As sex cannot be determined externally for juvenile sea turtles and there are little data on growth of loggerhead turtles of known sex in the Western North Atlantic we could not attempt to look at sex-specific growth rates.

The new growth curve is derived from animals throughout the southeast U.S. and cannot be used to address the question of regional variability in growth rates. But the intrinsic rate of growth (k) for this curve did not deviate much from that calculated by Braun-McNeill *et al.* (in prep) and is comparable to those estimated by Schmid (1995) and Foster (1994) (Table 8, Fig. 7). Hence we feel it is the best overall representation of loggerhead growth rates for the southeast U.S. available to date and we use it in the current model to estimate stage-durations, time-to-maturity and age-at-size.

The Frazer (1987) curve was prepared from juvenile growth rates of wild loggerheads in Florida that had overall higher growth rates than those measured in North Carolina (Mendonca 1981, Frazer and Ehrhart 1985, Frazer 1987, Braun-McNeill *et al.* in prep) and we cannot discount the possibility that this curve is representative of a maximum growth rate for wild loggerheads. Because of this and to be consistent with the previous models, we also consider models based on Frazer's (1987) growth curve.

Bjorndal *et al.* (2000) evaluated the duration of the pelagic stage. Their results estimate a minimum time of 6.5 years and an average time of 8 years for the duration of this stage. As the model we are using incorporates the first year into the fecundity function, we use 6 years and 7 years as minimum and average durations of the pelagic stage.

²¹ Unpublished Data. The Cooperative Marine Turtle Program was established by NMFS in 1980 to centralize the tagging programs among sea turtle researchers, distribute tags, manage tagging data, and facilitate the exchange of tag data.

Size-at-Stage

The earlier models define 58.1 cm SCL as the break between small and large juveniles and 87 cm SCL as the size at maturity (Crouse *et al.* 1987, Crowder *et al.* 1994). TEWG (1998) recommends 92 cm SCL as the average size of neophyte nesters. Heppell *et al.* (in press) uses 45 cm SCL as size at first settlement from pelagic to benthic habitats and 92 cm SCL as size at maturity.

In the models used for this stock assessment, we consider two size-at-stage scenarios. The first looks at a minimum size-to-stage and the second an average size-to-stage. Bjorndal *et al.* (2000) suggests 42 cm SCL as the smallest size at first settlement for loggerheads. Bjorndal *et al.* (2000) also estimate that the average size at settlement is 53 cm CCL or 49cm SCL (using their SCL to CCL conversion equation).

For size at maturity, we analyzed the CMTTP²¹ for original tagging events from nesting beach survey projects where SCL was recorded. We calculated an average of 90.38 cm SCL (SD=5.08) with the 5th and 95th percentiles equal to 82.5 and 99.2 cm SCL respectively (Fig 10). Given that some individuals might nest before they get tagged for the first time or the first tag might have been lost and the turtle not recognized as having been tagged, we acknowledge that 90.38 cm SCL is perhaps biased large as an average size-to maturity.

Hence, for the minimum size-to-stage scenario we use 42 cm SCL as the cutoff between pelagic juveniles and small benthic juveniles and 83 cm SCL (from the 5th percentile of the analysis of the CMTTP)²¹.as size-to-maturity. For the average size-to-stage scenario we use Bjorndal *et al.*'s (2000) estimate of 49 cm SCL as the cutoff between pelagic juveniles and small benthic juveniles and 90 cm SCL (calculated from the CMTTP²¹) for average size-to-maturity.

Sex Ratios

The sex of loggerhead sea turtle hatchlings is environmentally determined by a restricted range of nest incubation temperatures. Pivotal and transitional ranges of temperatures determine if the nest will produce males, females or both (Mrosovsky and Pieau 1991). Mrosovsky and Provancha (1989) suggest that the majority of a major rookery near Cape Canaveral, Florida incubates at such warm temperatures that virtually no males are produced. Presumably because of a shorter nesting season, characterized by cool beginning and ending temperatures, males are predominately produced in the Northern subpopulation.

We assessed the sex ratios of benthic loggerhead sea turtles by analyzing the STSSN database²² for dead-stranded loggerheads for which sex had been ascertained by direct examination of the gonads. It is likely that adult loggerheads have sex specific dispersal and consideration of adults in the analyses may bias the results. Therefore, to be conservative we

²² The Sea Turtle Stranding and Salvage Network is a cooperative endeavor between NMFS, other federal agencies, the states, many academic and private entities, and innumerable volunteers. Data are archived at the National Marine Fisheries Service Southeast Fisheries Science Center in Miami, Fla.

only considered loggerheads less than 80 cm SCL in order to eliminate adults from the analysis. Sex ratios were then assessed by statistical zone and by state (Table 9).

From mtDNA analyses, we know that the feeding aggregations of juvenile loggerheads are composed of turtles from the different subpopulations. Bass *et al.* (1999²³) analyzed genetic samples taken from stranded animals from 5 states, Texas, Florida, South Carolina, North Carolina and Virginia (Fig. 1). We combined information regarding the sex ratios of the juvenile feeding aggregations with the natal origin probabilities to determine the sex ratios specific to the analyzed subpopulations.

We restricted our analysis to states where sample sizes were sufficiently large ($N \geq 100$), where samples could be definitely assigned to relatively small (<500 km) geographic areas, and where all samples were analyzed for the same suite of contributing source populations. Data on Florida was not included because it did not meet the small geographic area criteria defined above. The sample size from Virginia was too small ($N=35$). The sample size from North Carolina also was small ($N=60$), however in another study, additional North Carolina samples were analyzed, increasing the sample size to 286 (Bass *et al.* 2000²⁴).

We used the genetics data from Texas ($N=121$)²³, South Carolina ($N=95$)²³ and North Carolina ($N=286$)²⁴ in combination with juvenile sex ratios from those states (Table 9) to set up the following linear equations:

$$\begin{aligned} 74.21 &= 83.36\mathbf{S} + 10.33\mathbf{N} + 6.30\mathbf{M} \quad (\text{TX sex ratio and natal origin probabilities}^{23}) \\ 67.44 &= 65.66\mathbf{S} + 24.55\mathbf{N} + 9.77\mathbf{M} \quad (\text{SC sex ratio and natal origin probabilities}^{23}) \\ 65.25 &= 64.04\mathbf{S} + 29.78\mathbf{N} + 5.82\mathbf{M} \quad (\text{NC sex ratio and natal origin probabilities}^{24}) \end{aligned}$$

S is the percent female hatchlings produced by the South Florida subpopulation, **N** is the percent female produced by northern subpopulation and **M** is the percent female hatchlings from the Yucatán subpopulation. The above three equations in three unknowns solved to give the following percentages:

S = 80% Female
N = 35% Female
M = 69% Female

We can estimate the south Florida subpopulation produces 80% females and the northern subpopulation produces 65% males. Limited data for the Yucatán subpopulation suggest nearly 70% of hatchlings are female. The sex ratios for the northern and south Florida subpopulations are consistent with what is known about the temperature-dependent sex determination of

²³ Bass, A.I. S-M. Chow, and B.W. Bowen. 1999. Final report for project titled: genetic identities of loggerhead turtles stranded in the Southeast United States. Unpublished report to National Marine Fisheries Service, order number 40AANF809090. Department of Fisheries and Aquatic Sciences, University of Florida, Gainesville, Fla., 11pp.

²⁴ Bass, A.L. S.P. Epperly, J. Braun-McNeill and A. Francisco. 2000. Temporal variation in the composition of a loggerhead turtle (*Caretta caretta*) developmental habitat. Unpublished manuscript. Department of Fisheries and Aquatic Sciences, University of Florida, Gainesville, Fla., 26 pp.

loggerheads. For lack of specific data to the contrary, previous models have used 0.5 as the default sex ratio for loggerheads (Crouse *et al.* 1987, Crowder *et al.* 1994 and Heppell *et al.* in press). We now have regional sex ratios to use in the model but also construct the same models with a sex ratio of 0.50 for comparison with historical models.

Survival Rates

For the model runs in which stage duration was estimated using Frazer's (1987) growth curve, we use the same survival rates that were estimated by Frazer (1983a, 1986) and used in the previous models (Crouse *et al.* 1987, Crowder *et al.* 1994, Heppell *et al.* in press). Heppell *et al.* (in press) found they needed to increase survival rates from the previous models to gain a realistic life history for the longer stage durations. Hence we wanted to readdress benthic juvenile and adult survival rates.

Benthic Juveniles

Frazer (1987) estimated juvenile survival rates for loggerhead sea turtles using a catch curve (Seber 1982). But it seems likely that if a faster growth curve is used to estimate age-at-size, the resulting slope on the catch-curve will be steeper than for age-at-size calculated from slower growth curves. Steeper slopes correlate with higher instantaneous mortalities.

We analyzed the STSSN²² data using a catch curve analysis. We used only data from 1986-1989 (pre-TED), assuming the population was at a stable age distribution at that time and that dead stranded animals are a representative cross-section of the body sizes of turtles in the population. Catch curves are created by plotting $\ln(N_x)$ versus x where x is age and N_x is the number of individuals in the sampled population at age x . The age at which all individuals have fully recruited to the population (threshold age (Seber 1982)) is estimated as the peak in the curve. Age-at-size was calculated for each dead stranded loggerhead using the new growth curve. N_x was calculated for each one-year age class (x) and $\ln(N_x)$ was plotted versus x (Fig. 11).

Threshold age was determined at 2 years post-settlement. We calculated the instantaneous mortality rate (z) from linear regressions on the declining arm of the catch curve in three different ways (Fig. 12):

- From threshold age to the age corresponding to 70 cm SCL
- From threshold age to the age corresponding to 90 cm SCL
- From threshold age-to-age 20, the point where the trend in the curve start to break-up.

Breaking the catch curve at the 3 different points resulted in similar survival rates, 0.893, 0.929 and 0.908 respectively. To be conservative, we use 0.893 as the pre-TED annual survival rate for small and large benthic juveniles in the current models.

TEWG (2000, pg. 46) reviews estimates of quantitative decreases in strandings after the imposition of TED regulations based on analyses of strandings from South Carolina and Georgia

(Crowder *et al.* 1995, Royle and Crowder 1998²⁵, Royle 2000²⁶). It is unclear how to relate the percent decreases in strandings (reported at between 37 and 58%) to reductions in instantaneous mortality (z). Heppell *et al.* (in press) used a value of 30% reduction in mortality as the amount by which TED use reduces overall mortality of the affected stages of loggerhead sea turtles.

As the smallest TED openings only allow small turtles to pass through¹⁹, we applied a 30% reduction in mortality to small benthic juveniles only to estimate the annual survival rate of this size class after 1990. We calculated the new annual survival rate for small benthic juveniles by multiplying z , the instantaneous rate of mortality by 0.7.

Adults

For adult survival probabilities, we analyzed nesting beach tag return data for two nest monitoring projects using modifications of the Cormack-Jolly-Seber approach (Cormack 1964, Jolly 1965, Seber 1965, Lebreton *et al.* 1992). The two nesting beaches analyzed were Melbourne Beach, Florida²⁷ and Wassaw Island, Georgia²⁸ (Williams and Frick 2001). For Wassaw Island, tag-loss was accounted for according to Frazer (1983b). The program MARK (White and Burnham 1999) was used to estimate survival rates for each data set based on the models of Lebreton *et al.* (1992). For Wassaw Island, the model incorporating time dependent survival probability (ϕ_t) and time independent capture probability (p) gave the best goodness-of-fit based on Akaike's Information Criterion. For Melbourne Beach the model giving the best goodness-of-fit incorporates time dependent survival and capture probabilities (ϕ_t, p_t). Average estimated annual survival probabilities were 0.79 for Wassaw Island and 0.83 for Melbourne Beach. The models used do not account for emigration, hence,

$$\phi_t = S_t(1-E_t)$$

where S_t is the annual survival rate and E_t is the emigration rate. We know that nesting loggerheads do not have strict nest-site fidelity (CMTTP²¹, LeBuff 1974, Ehrhart 1979², Bjorndal *et al.* 1983, LeBuff 1990) but the actual value of E_t is unknown so we use ϕ_t as an estimate of annual survival acknowledging that this value is lower than the true survival rate and therefore conservative. The annual survival rates calculated from the tag-return data (0.79 and 0.83) are in close agreement with the value of 0.8091 originally estimated from Little Cumberland Island data (Frazer 1983b). In light of the uncertainty associated with these values, we selected the mean of all three values, 0.812 as representative of adult annual survival in the current model.

²⁵ Royle, J.A. and L.B. Crowder. 1998. Estimation of a TED effect from loggerhead strandings in South Carolina and Georgia strandings data from 1980-97. Unpublished report, U.S. Fish and Wildlife Service, Laurel, Maryland, 11pp.

²⁶ Royle, J.A. 2000. Estimation of the TED effect in Georgia shrimp strandings data. Unpublished report, U.S. Fish and Wildlife Service, Laurel, Maryland, 11pp.

²⁷ Ehrhart, L.M. Unpublished data. Department of Biology, University of Central Florida, Orlando, Fla.

²⁸ The Caretta Research Project, Savannah Science Museum, P.O. Box 9841 Savannah Ga. and the U.S. Fish and Wildlife Service, Savannah Coastal Refuges, 1000 Business Center Drive, Suite 10, Savannah, Ga.

Pelagic Juveniles

Due to the cryptic nature of this life stage, no data are available to directly measure pelagic juvenile survival rates. Because we have estimates for all other inputs into the model, we can infer pelagic survival rates from those rates and population trends. We assessed the range in potential annual survival rates of pelagic juveniles by allowing for the uncertainties in other parameter estimates and running the model using combinations of the inputs as discussed in the previous sections and three values for λ for the northern subpopulation ($\lambda = 0.95, 0.97$ and 1.0) as discussed above in the Status and Trends section. We feel that the overall population trend for the northern subpopulation prior to 1990 is encompassed within the range of λ values we used.

Fecundity

Heppell *et al.* (in press) used reproduction parameters from TEWG (1998) and survival to year 1 from Frazer (1983). We use the same values for the current models, which are nests per breeding female = 4.1, eggs per nest = 115, and survival to year 1 = 0.6747. The fecundity value in the matrix is:

$$F = 4.1 \times 115 \times (\text{proportion of female offspring}) \times 0.6747.$$

Population Models

We considered four different stage duration scenarios (Models 1-4). These were based on the two individual growth models, Frazer's (1987) (Frazer) and the new one presented here (New). For each growth curve, we estimated stage durations based on the minimum-size-to-stage and the average-size-to-stage values discussed in the size-at-stage section and survival rates were used as discussed previously (Tables 10-13). We used the same fecundity parameters as in Heppell *et al.* (in press) with the exception of the sex ratio.

For each model, we ran 3 scenarios, using $\lambda = 0.95, 0.97$ and 1.00 . As these reflect the range of estimates for the pre-1990 population growth rates for the northern subpopulation, we used 0.35 as the proportion of female offspring in these models. For each of these 12 (4 models times 3 population growth rates) runs of the model, we determined the appropriate annual survival rate for the pelagic stage (Table 14). In Model 2, the pelagic annual survival probability for the $\lambda = 1.0$ scenario would have to have exceeded 1.0, so we discount this possible combination of vital rates and consider only the remaining 11 runs of the model.

The right eigenvector of a projection matrix gives the proportional distribution of ages for a population at a stable age distribution (Caswell 2001). To check how well the age distributions associated with each model correlates with the natural population, we summed the proportional contributions across the benthic stages (small, large, and adult) to get the predicted stable stage structure. We compared this to the observed stage structure based on an analysis of strandings between 1986 and 1989 (the same data used to create the catch curve) (Fig. 13). Models 3 and 4 appear to have the best fit with the strandings data.

Elasticity of Stages

For matrix projection models, an elasticity analysis examines the proportional contribution of the asymptotic population growth rate (λ) to changes in the vital rates that compose the elements of the transition matrix (de Kroon *et al.* 1986, 2000). Elasticities also reveal the proportional contribution of each element of the matrix to λ . For an age-based matrix, elasticities can be summed over stages to find the proportional contribution of each major life-stage to λ . The elasticity of λ to juvenile stage is dependant on the duration of those stages (Caswell 2001, Heppell *et al.* 2000). Longer stage lengths have higher elasticities. Thus, for Model 1, small and large benthic juveniles have the same elasticity (Fig. 14). For Model 2, small benthic juvenile elasticity is lower than that of pelagic juveniles while the elasticity of the large benthic juveniles is much higher than either of the other juvenile stages. Similarly, the elasticities of the juvenile stages for Models 3 and 4 correlate with the stage durations (Tables 12 and 13) and the longest stage duration, the large benthic juvenile stage of Model 4, has the highest elasticity (Fig. 14). These are the elasticities for $\lambda = 0.95$, the specific values change only slightly with changes in λ and the overall trends remain the same.

Sex Ratios

There is no reason to expect different pelagic juvenile stage survival rates for loggerheads originating from the south Florida subpopulation as compared to the northern subpopulation. For the benthic stages, there are potential differences in nearshore mortality from anthropogenic sources. As we have no current means of quantifying such differences, we assume the benthic stage survival rates are the same for both subpopulations. There is, however, evidence of a higher proportion of females being produced in the South Florida subpopulation. Hence, we also ran the same 11 models as described previously, with a proportion of female offspring equal to 0.80. For consistency with the historical models, we also ran the 11 models with a proportion of female offspring equal to 0.50.

Population Projection

Following Heppell *et al.* (in press), post-1990 population trajectories were run for each model (now numbering 33 – 11 times the 3 sex ratios) by initializing with a population at stable age distribution for the appropriate combination of model and λ , assuming 2000 nesting females (TEWG 1998). Small benthic juvenile mortality was decreased by 30% and the population projected based on the new survival rates (Fig. 15-17). Obviously, increasing small benthic juvenile survival rates has the effect of increasing population growth rates for each model scenario (see Fig. 18 for new population growth rates). However, when the populations are initialized at a declining rate of 5% per year ($\lambda=0.95$), a 30% decrease in mortality of small benthic juveniles is not enough to reverse the declining trends regardless of the sex ratio (Fig. 18). At an initial population decline of 3% per year, declining trends are reversed in Models 1 and 2 except at a sex ratio of 0.35 for Model 2. At stable population growth, $\lambda=1.0$, a 30% decrease in small benthic juvenile mortality alone results in increasing population trends in all model scenarios (Fig. 18). Note that the λ values given in Fig. 15-18 are the initial population growth rates. The populations in these projections will eventually stabilize to the respective population growth rates indicated in Fig. 18.

TEWG (1998) presented a population model for the Kemp's ridley sea turtle for which the model projections were fit to observed nesting trends. This was possible for Kemp's as there is only a single stock with one primary nesting aggregation and 30 years of nesting trend and hatchling production data. Loggerheads of the southeast U.S. have a much more complicated stock structure with numerous nesting aggregations, only some of which are currently monitored and very few were regularly monitored prior to 1989.

We are using the four model scenarios, each with three starting λ values to address the uncertainties in the model parameters. The actual stage duration and population growth rates are likely bracketed. Due to the uncertainty inherent in these models, we do not assert that the population projections presented here and elsewhere in this document are quantitative predictions of future sea turtle numbers. They should be viewed only as qualitative outcomes of the implementation of management strategies (or lack thereof), indicating the time lags that can be expected before the effects of management are seen in terms of numbers of nesting females (Crowder *et al.* 1994). This is also why we do not put specific years on the x-axis of the projection plots (Fig. 15-17).

We start the population projections at stable age distribution. At time one we increase survival of the small benthic juveniles which perturbs the population out of stable age distribution, giving a pulse of small benthic juveniles. The lag time before the initial pulse of small benthic juveniles are seen as an increase in the number of nesting females is equal to the length of the duration of the large benthic juvenile stage. After a length of time equivalent to the duration of the small benthic juvenile stage, this pulse in the numbers of nesting females levels out and the populations temporarily stabilize. However, there are now increased numbers of nesting females producing increased numbers of offspring. Following a period equal to age at reproductive maturity, when these increased numbers of offspring begin to mature, another pulse is observed in the number of nesting females. Due to the duration of the stages, the latter pulse is seen only in Model 1. For this model the duration of the large benthic juvenile stage is 7 years. The first pulse for Model 1 occurs at 7 years (Fig. 15-17, Model 1). The duration of the small benthic juvenile stage is 7 years, hence, after 14 years the initial pulse levels out (Fig. 15-17, Model 1). Age to reproductive maturity for Model 1 is 21 years, therefore, 21 years after the first pulse began, or at 28 years, the number of nesting females pulses again. The pulses will continue until the populations again reach stable age distribution, which often takes two generations or more. Similar dynamics are occurring in the population projections for the other models, however, the time series were not run long enough to see the effects of increased numbers of offspring (Fig. 15-17; Models 2-4).

In model 2, the populations are still declining for 35% female offspring and starting $\lambda = 0.95$ and 0.97 , and for 50% female offspring and starting $\lambda = 0.95$ following the increase in small benthic juvenile survival (Fig. 18). The populations are slightly increasing following the increase in small benthic juvenile survival for starting $\lambda = 0.97$ (Fig. 18). As described above, there is a surge in number of nesting females as the increased numbers of small benthic juveniles pulse through, after which the population continues to decline (Fig. 15-17; Model 2). For Models 3 and 4, $\lambda = 0.95$ and 0.97 , similar dynamics are occurring, however, the length of the large benthic juvenile stage is very long and the populations are still declining by as much as 4%

per year (Fig 15-17, Models 3 and 4; Fig. 18), hence the pulse of small benthic juveniles is not as obvious.

The meta-analysis of nesting trends for the northern subpopulation indicates that numbers of nesting females in this region may have increased since 1990. In our models, we only allow for increases in small benthic juvenile survival and thus it takes a period equal to the duration of the large benthic juvenile stage to begin to see increases in numbers of nesting females (Fig. 15-17, Tables 10-13). The effects of TED use on decreasing mortality in sea turtles have been documented quantitatively (TEWG 1998, Crowder *et al.* 1995, Royle and Crowder 1998²⁵, Royle 2000²⁶). Using the cut-off of 70 cm SCL and below for the benefits of TED use is also justified as that is about the maximum size turtle that can fit through the smallest size TED openings allowed under current regulations (Epperly and Teas 1999¹⁹).

There are other anthropogenic sources of sea turtle mortality that have been mitigated over the years. For example, when the loggerhead sea turtle was listed as a threatened species in 1978 under the Endangered Species Act of 1973 (PL93-205), taking eggs and nesting females, and keeping in-water catches became illegal. South Carolina sturgeon fishers using large mesh gill nets and operating in the coastal waters of South Carolina and North Carolina, were implicated in mass-dead-stranding events of loggerheads up to 89 cm SCL from mid-April to early May of 1977 and 1981 (Crouse 1985, Ulrich 1978²⁹). This fishery was closed in 1986 in South Carolina (NMFS and USFWS 1991) due to declines in sturgeon populations. North Carolina initially imposed restrictions on the use of large mesh gill nets between February and September (N.C. Marine Fisheries Regulations, NCAC 15 3B.0402(5)) and as of 1991, the sturgeon fishery has been closed. The state of Florida now prohibits the use of entangling nets (Florida Fish and Wildlife Conservation Commission, Division of Marine Fisheries Regulations, Chapter 68B-4.0081, issued 3-1-92, amended, 7-18-94 and 4-27-98). Takes of pelagic juvenile loggerheads in US and international longline fisheries in the North Atlantic are only now being quantified, but estimates from the Eastern North Atlantic are large (Bolten *et al.* 1994) and could alter population trends (Crowder *et al.* 1994).

Combining these factors and possibly others that are not documented may contribute to the potentially increasing trends in nesting females seen in the meta-analysis results for the northern subpopulation, but that analysis is presented with caution as it is unweighted and does not consider the relative abundance of each beach. As factors may have combined to contribute to possibly increasing nesting population trends for the northern subpopulation, they would be accounted for in the scenarios that set $\lambda = 1.0$. Conversely, there are likely other sources of mortality offsetting the mitigated ones that are resulting in the slow-down of increasing nesting trends in the south Florida subpopulation. None of these other mortality sources are well studied or documented and cannot be considered quantitatively in the population models.

There is some concern about the nest trend data used in the meta-analysis. It is possible that what appears to be increasing trends is an artifact of increasing survey efforts. Attempts were made to circumvent this possibility by only using data that appeared to represent consistent

²⁹ Ulrich, G.F. 1978. Incidental catch of loggerhead turtles by South Carolina commercial fisheries. Unpublished report to National Marine Fisheries Service, contract numbers 03-7-042-35151 and 03-7-042-35121. South Carolina Wildlife and Marine Resources Department, Charleston, S.C. 36pp.

effort, however, we also do not want to overestimate population growth rates for loggerheads. Therefore, we continue to consider all three possible scenarios in the impact assessment. We also need to consider that nesting trends reflect trends in only a very small portion of the overall population and that uncertainties not included in the model do not provide assurance that populations will recover.

For $\lambda=0.97$ (the median λ evaluated) the models based on the new individual growth curve, Models 3 and 4, using sex ratios of 0.35 or 0.50, all suggest declining populations after a 30% reduction in mortality for small benthic juveniles. At a sex ratio of 0.8, the population growth rates were positive for all models except for Model 4. For the sex ratio representative of the northern subpopulation, 0.35, a 30% decrease in mortality for small benthic juveniles was not enough to stabilize the population growth rate unless the initial $\lambda=1.0$.

Impacts on the Populations

Recent Stranding Events and Trends

From 1998-2000, strandings decreased in the traditionally high zones 28-32 along the Atlantic coast (Table 15)¹². Strandings in the mid-Atlantic zones 35-37 continued to show an increasing trend, with loggerhead strandings in zone 35 reaching an unprecedented total of 396 in 2000. More than half of these turtles washed ashore during April and the first week of May and were likely due to large-mesh gillnet fisheries operating in the area (65 FR 31500-31503, May 18, 2000).

Strandings along the southern Florida Gulf coast and in the Florida Keys were approximately double historic levels in 2000. A persistent red tide during the first five months of the year³⁰ may have played a role in the increased strandings, especially in zone 3. Loggerhead strandings in southwest Florida were elevated throughout the shrimping season, possibly as a result of the turtles being too large to fit through the current TED openings (Epperly and Teas 1999³¹). Beginning in October, many large loggerheads have been found floating with an illness of undetermined cause in southern Florida and the Keys. These turtles all are extremely weak; they cannot lift their heads out of water to breathe and most have developed secondary pneumonia due to aspiration of water into the lungs³². The mortality rate for turtles found alive with these symptoms has been greater than 50% and the turtles that are still alive in rehabilitation facilities are showing few signs of improvement. Researchers believe the turtles may be suffering from a toxin (*Ibid.*).

³⁰ Mote Marine Laboratory, Sarasota, Fla. 2000. Red Tide Chronology. www.mote.org/~mhenry/rtchrono.phtml

³¹ Epperly, S.P. and W.G. Teas. 1999. Evaluation of TED opening dimensions relative to size of turtles stranding in the western North Atlantic. Unpublished Report. NMFS SEFSC Contribution PRD-98/99-08, National Marine Fisheries Service, SEFSC, Miami, Fla., 31 pp.

³² Richie Moretti, Sea Turtle Hospital, Marathon, FL. Personal Communication (phone) to Sheryan Epperly, National Marine Fisheries Service, SEFSC, Miami, Fla., January 17, 2001.

Fate of Dead Turtles

In FY96 NMFS Office of Protected Resources contracted with Duke University Marine Laboratory to study the fate of turtles dying at sea to better understand what numbers of stranded turtles represent. The results will be presented at the upcoming 21st Annual Symposium on Sea Turtle Biology and Conservation in Philadelphia, Pennsylvania in February (P. Mooreside personal communication³³). Fifteen years of hourly wind speed data, recorded off the North Carolina coast, were transformed into vectors, converted into wind stress magnitude and direction values, and averaged by month. Near-shore surface currents were then modeled for the South Atlantic Bight via a three-dimensional physical oceanographic model (Werner *et al.* 1999). Estimated water currents and particle tracks were compared to the spatial locations of sea turtle carcasses stranded along ocean-facing beaches of North Carolina. On average, the number of carcasses stranded on ocean-facing beaches may represent, at best, approximately 20% of the total number of available carcasses at-sea. This evidence, in accordance with the spatial behavior of modeled lagrangian drogues, indicates that only those turtles killed very close to the shore may be most likely to strand.

Anthropogenic Impacts

A number of anthropogenic impacts have been identified for loggerhead sea turtles (National Research Council 1990, NMFS & USFWS 1991) 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 has come to light concerning 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 TEWG reports (1998, 2000). Known sources of impact are listed in Appendix 2.

Pelagic longline fisheries
See Part III.

Trawls

A detailed summary of the U. S. shrimp trawl fishery and the Mid-Atlantic winter trawl fishery impacts can be found in the TEWG reports (1998, 2000). Other bottom trawl fisheries that are suspect for the incidental capture of sea turtles are the horseshoe crab fishery in Delaware (Spotila *et al.* 1998⁴⁶) and the whelk trawl fishery in South Carolina (Sally Murphy personal communication³⁴) and Georgia (Mark Dodd personal communication³⁵). In South Carolina, the whelk trawling season opens in late winter and early spring when offshore bottom waters are > 55°F. One criterion for closure of this fishery is water temperature: whelk trawling closes for the season and does not reopen throughout the State 6 days after water temperatures first reach 64°F in the Fort Johnson boat slip. Based on the South Carolina Department of Natural Resources Office of Fisheries Management data, approximately 6 days will usually lapse before water temperatures reach 68°F, the temperature at which sea turtles move into State waters (David Cupka personal communication³⁶). From 1996-1997, observers onboard whelk

³³ Pete Mooreside, Duke University Marine Laboratory. Personal Communication (E-Mail of draft extended abstract) to Sheryan Epperly, National Marine Fisheries Service, SEFSC, Miami, Fla., January 23, 2001.

trawlers in Georgia reported a total of 3 Kemp's ridley, 2 green and 2 loggerhead sea turtles captured in 28 tows for a CPUE of 0.3097 turtles/100ft net hour³⁵. As of December 2000, TEDS are required in Georgia state waters when trawling for whelk (*Ibid.*).

A loggerhead was reported captured in a Florida try net (W. Teas personal communication³⁷). Shrimp trawlers operating in the waters off Venezuela were reported to have captured a total of 48 sea turtles, of which 15 were loggerheads, from 13, 6000 trawls (Marcano and Alio 2000). They estimated annual capture of all sea turtle species to be 1370 with an associated mortality of 260 turtles.

Gill nets

A detailed summary of the gill net fisheries currently operating along the mid- and southeast U.S. Atlantic coastline that are known to incidentally capture loggerhead can be found in the TEWG reports (1998, 2000). 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 Rhode Island, Connecticut, New York, New Jersey, Delaware, Maryland, Virginia, and North Carolina state waters and/or federal waters offshore thereof. Incidental captures in these gill net fisheries (both lethal and non-lethal) of loggerhead, leatherback, green and Kemp's ridley sea turtles have been reported (W. Teas, personal communication³⁷, J. Braun-McNeill personal communication³⁸). In addition, illegal gill net incidental captures have been reported in South Carolina, Florida, Louisiana and Texas (W. Teas personal communication³⁷). See Appendix 2 for additional information.

On October 27, 2000, the North Carolina Division of Marine Fisheries (NCDMF) closed waters in the southeastern portion of the Pamlico Sound to commercial large-mesh flounder gill netting as a result of elevated turtle takes by the fishery. From September 15–October 25, observers documented 17 gill net interactions, eight of which were loggerheads (six released

³⁴ Sally Murphy, South Carolina Department of Natural Resources, Charleston, S.C. Personal Communication. (Phone) to J. Braun-McNeill, National Marine Fisheries Service, SEFSC, Beaufort, N.C., November 27, 2000.

³⁵ Mark Dodd, Georgia Department of Natural Resources, Brunswick, Ga. Personal Communication (Fax) to Joanne Braun-McNeill, National Marine Fisheries Service, SEFSC, Beaufort, N.C., December 8, 2000.

³⁶ David Cupka, South Carolina Department of Natural Resources, Marine Resources Division, Charleston, S.C. Personal Communication (E-Mail of the Management Plan for South Carolina's Offshore Whelk Trawling Fishery - updated January 1999) to Joanne Braun-McNeill, National Marine Fisheries Service, SEFSC, Beaufort, N.C., December 18, 2000.

³⁷ Wendy Teas, National Marine Fisheries Service, SEFSC, Miami, Fla. Unpublished STSSN strandings data. Personal Communication (E-Mail of strandings data) to Joanne Braun-McNeill, National Marine Fisheries Service, SEFSC, Beaufort, N.C., December 12, 2000.

³⁸ Unpublished Data. Joanne Braun-McNeill, National Marine Fisheries Service, SEFSC, Beaufort, N.C. Personal Communication, December 21, 2000.

alive and two dead or injured³⁹). There also were 15 strandings documented from nearby areas during the same time period.⁴⁰ The NCDMF and NMFS had just agreed on details of a Section 10 Permit to the Endangered Species Act for the flounder fishery just prior to the closure⁴¹. The permit established allowable levels of live and lethal gill net interactions for each turtle species, with a goal of reducing strandings by at least 50 percent from 1999 levels. The fishery was closed when the incidental take level was met for green sea turtles⁴². The NCDMF estimated that there were 50 loggerheads captured at the time of closure and that 44 of those had been drowned³⁹.

From 1981-1990, 397 loggerhead sea turtles were incidentally captured in gill nets set by Italian fishermen in the central Mediterranean Sea; gill net mortality was reported to be 73.6% (Argano *et al.* 1992). An additional study in this same area estimated 16,000 loggerheads/year are captured by net with 30% mortality (De Metro and Megalfonou 1988). Observers of the Spanish driftnet fishery in the western Mediterranean documented the incidental capture of 30 loggerheads from 1993-1994, of which one was dead; an estimated 236 loggerheads were caught in 1994 (Silvani *et al.* 1999). In Nicaragua, although green and hawksbill turtles are targeted, loggerhead and leatherback turtles are incidentally caught by gill net (Lagueux 1998, Lagueux *et al.* 1998, Lima *et al.* 1999); an estimated 600 loggerheads are caught each year (Lagueux 1998). Gill nets set for finfish and sharks in Belize are also suspected of catching sea turtles (Smith *et al.* 1992). Of the 500-800 turtles sold annually in Belize, 30% are reported to be loggerheads (*Ibid.*).

Hook and line

Loggerheads are known to bite a baited hook, frequently ingesting the hook. Hooked turtles have been reported by the public fishing from boats, piers, beach, banks, and jetties (Cannon *et al.* 1994, J. Braun-McNeill personal communication³⁸, A. Cannon personal communication⁴³, Spotila *et al.* 1998⁴⁴, STSSN unpublished data¹²) and from commercial fishermen fishing for reef fish and for sharks with both single rigs and bottom longlines (S.

³⁹ Excel spreadsheet as attachment to E-Mail from Jeff Gearhart, N.C. Division of Marine Fisheries, Morehead City, N.C. to David Bernhard, National Marine Fisheries Service, SERO, St. Petersburg, Fla., October 25, 2000.

⁴⁰ North Carolina Division of Marine Fisheries news release, NR-61-2000, "Commercial Flounder Season Closes to Protect Sea Turtles", Morehead City, N.C., October 25, 2000.

⁴¹ National Marine Fisheries Service. Endangered Species Act Section 10 Permit #1259 issued to State of North Carolina, Department of Environmental and Natural Resources, Division of Marine Fisheries, Morehead City, N.C., October 5, 2000.

⁴² North Carolina Division of Marine Fisheries news release, NR-61-2000, "Commercial Flounder Season Closes to Protect Sea Turtles", Morehead City, N.C., October 25, 2000.

⁴³ Andrea Cannon, National Marine Fisheries Service, SEFSC, Galveston, Texas. Personal Communication to Sheryan Epperly, National Marine Fisheries Service, SEFSC, Miami, Fla.

⁴⁴ Spotila, J.R., P.T. Plotkin, and J.A. Keinath. 1998. In water population survey of sea turtles of Delaware Bay. Unpublished Report. Final Report to NMFS, Office of Protected Resources for Work Conducted Under Contract #43AANF600211 and NMFS Permit No. 1007 by Drexel University, Philadelphia, Penn., 21 pp.

Epperly personal communication⁴⁵). A detailed summary of the impact of hook and line incidental captures to loggerhead sea turtles can be found in the TEWG reports (1998, 2000).

Power plants

Power plants are known to entrain loggerhead sea turtles at the intake canals to their cooling systems. A detailed summary of the incidental capture of loggerhead sea turtles in power plant intake screens can be found in the TEWG reports (1998, 2000).

Pound Nets

Pound nets are a passive, stationary gear that are known to incidentally capture loggerhead sea turtles in Massachusetts (R. Prescott personal communication⁴⁶), Rhode Island, New Jersey, Maryland (W. Teas personal communication³⁷), New York (Morreale and Standora 1998), Virginia (Bellmund *et al.*, 1987) and North Carolina (Epperly *et al.* 2000). Although pound nets are not a significant source of mortality for loggerheads in New York (Morreale and Standora 1998) and North Carolina (Epperly *et al.* 2000), they have been implicated in the deaths of loggerheads in the Chesapeake Bay from mid-May through early June (Bellmund *et al.* 1987). The turtles were reported entangled in the large mesh (>8 inches) pound net leads.

Other Fisheries

Incidental captures of loggerheads in fish traps set in Massachusetts, Rhode Island, New York, and Florida have been reported (W. Teas personal communication³⁷). Although no incidental captures have been documented from fish traps set in North Carolina⁴⁷ and Delaware (Anonymous 1995⁴⁸), they are another potential anthropogenic impact to loggerheads and other sea turtles. Lobster pot fisheries are prosecuted in Massachusetts (Prescott 1988), Rhode Island (Anonymous 1995⁴⁸), Connecticut (*Ibid.*) and New York (S. Sadove personal communication⁴⁹). Although they are more likely to entangle leatherback sea turtles, lobster pots set in New York are also known to entangle loggerhead sea turtles (*Ibid.*). No incidental capture data exist for the other states. Long haul seines and channel nets in North Carolina are known to incidentally capture loggerhead and other sea turtles in the sounds and other inshore waters (J. Braun-McNeill personal communication³⁸). No lethal takes have been reported. Whelk pots set in

⁴⁵ Sheryan Epperly, National Marine Fisheries Service, SEFSC, Beaufort, N.C. Personal Communication (discussions with commercial reef-fish and shark fishermen in North Carolina), 1984-1998.

⁴⁶ Robert Prescott, Massachusetts Audubon Society's Wellfleet Bay Wildlife Sanctuary, South Wellfleet, Mass. (E-Mail) to Joanne Braun-McNeill, National Marine Fisheries Service, SEFSC, Beaufort, N.C., December 1, 2000.

⁴⁷ Epperly, S.P. and V.G. Thayer. 1995. Marine mammal and sea turtle/fisheries interactions in North Carolina. Unpublished manuscript. National Marine Fisheries Service, SEFSC, Beaufort, N.C.

⁴⁸ Anonymous. 1995. State and federal fishery interactions with sea turtles in the mid-Atlantic area, p. 1-12. In Proceedings of the Workshop of the Management and Science Committee of the Atlantic States Marine Fisheries Commission July 17-18, Richmond, Virginia. Unpublished report of the ASMFC, Washington, D.C.

⁴⁹ Sam Sadove, Long Island University, Southampton College, Southampton, N.Y. Personal Communication (Phone) to Joanne Braun-McNeill, National Marine Fisheries Service, SEFSC, Beaufort, N.C., December 6, 2000.

Virginia and North Carolina could potentially entrap loggerheads as they attempt to get to the bait or the whelks within the trap (Mansfield and Musick 2000⁵⁰).

Bottom set lines in the coastal waters of Madeira, Portugal are reported to take an estimated 500 pelagic immature loggerheads each year (Dellinger and Encarnacao 2000). Adult female loggerheads are taken by hand by the indigenous people inhabiting Boavista Island, Cape Verde, Western Africa (Cabrera *et al.* 2000). In Cuba, loggerhead, along with green and hawksbill sea turtles, are commercially harvested (Gavilan 2000, Alvarez 2000).

Marine Debris

An additional source of mortality that has not been adequately assessed is the ingestion of anthropogenic debris by pelagic turtles. A summary of marine debris impacts can be found in the TEWG reports (1998, 2000).

Discussion

The recovery plan for this species (NMFS and USFWS 1991) states that southeastern U.S. loggerheads can be considered for delisting if, over a period of 25 years, adult female populations in Florida are increasing and there is a return to pre-listing annual nest numbers totaling 12,800 for North Carolina, South Carolina and Georgia combined (equates to approximately 3,100 nesting females per year at 4.1 nests per female per season). Nesting trends indicate the numbers of nesting females associated with the south Florida subpopulation are increasing. Likewise, nesting trend analyses indicate potentially increasing nest numbers in the northern subpopulation (TEWG 2000, Appendix 1). Given the uncertainties in survival rates discussed previously and the stochastic nature of populations, the population trajectories should not be used now to quantitatively assess when the northern population may achieve 3,100 nesting females.

Similar to results found in previous models, in all model scenarios presented herein, the juvenile stages have the highest elasticity and maintaining or decreasing current sources of mortality in those stages will have the greatest impact on maintaining or increasing current population growth rates. Again, these values are in direct proportion to the stage lengths determined from the individual growth models used, particularly for the model pairs that use the same survival rates (Models 1 and 2 and Models 3 and 4) (Heppell *et al.* 2000). We feel we have bracketed age-to-maturity with these model pairs, and, for the models using average age-to-maturity (Models 2 and 4), the elasticity of the large benthic juveniles are much higher than small benthic juveniles while the difference is not as pronounced in the minimum age-to-maturity models (Models 1 and 3). If the new individual growth model presented here accurately describes loggerhead growth rates and average size-to-maturity is around 90 cm SCL, large benthic juveniles greater than 70 cm SCL are a critical stage. This stage may not be fully

⁵⁰ Mansfield, K.L. and J. A. Musick. 2000. Characterization of the Chesapeake Bay pound net and whelk pot fisheries and their potential interactions with marine sea turtle species. Unpublished Report. Virginia Institute of Marine Sciences Interim Report to the National Marine Fisheries Service, Northeast Regional Office, Gloucester, Mass., 12 pp.

protected under current TED regulations. More information regarding growth rates, habitat utilization and related mortality sources specific to this stage is important.

As with the previous loggerhead models, the models presented herein assess females only and make the assumption that there are plenty of males in the population for maximum fecundity. The actual operational sex ratio necessary on the breeding grounds for maximum fecundity is unknown. In a genetic analysis of loggerhead clutch paternity, Moore (2000) found that eggs contained in 31% of the sampled nests reflected contributions from multiple fathers and 10% of the nests had 3 or more fathers. This degree of multiple paternity was detected by only sampling 10 eggs (<10%) per nest. She expressed concern that males may be a limiting factor at her study site as a previous study indicated >90% female hatchling production based on incubation temperatures (Mrosovsky and Provanha 1989).

New results from nuclear DNA analyses indicate that males do not show the same degree of site fidelity, as do females.⁹ It is possible, then, that the high proportion of males produced in the northern subpopulation are an important source of males throughout the southeast U.S., lending even more significance to the critical nature of this small subpopulation. Our current understanding of the loggerhead mating system is rudimentary, but further declines or loss of the northern nesting population (which produces a disproportionate share of males for the whole population) could contribute to a serious population decline over the entire region.

We have very little sex specific information on the vital rates of sea turtles. If males mature significantly faster than females and/or if males reproduce every year while females an average of every 2.5 years (Richardson and Richardson 1982), then the functional sex ratio will be very different from the actual sex ratio based on hatchling output. This would serve to alleviate the extreme female bias in hatchling production in Florida. Much more information is needed about the mating system of loggerheads and sex-specific vital rates in order to truly assess the impacts of the low production of males in the south Florida subpopulation.

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Table 1. Annual loggerhead turtle (*Caretta caretta*) nesting and hatching statistics, Dry Tortugas National Park, 1995-2000. Reproduced from Reardon (2000¹⁴).

Data Category	1995	1996	1997	1998	1999	2000
Total Nests	269	190	210	190	242	202
Nests Inventoried (n)	169	167	187	142	207	177
Average Clutch Size	98 eggs (50-188)	100 eggs (32-169)	105 eggs (48-169)	102 eggs (60-149)	103 eggs (30-162)	99 eggs (42-148)
Average Hatching Success*	82.00% (Not available)	78.80% (15.3-100.0%)	72.30% (0.0-100%)	76.30% (0.0-98.3%)	78.50% (0.0-99%)	74.90% (0.0-100.0%)
Average Incubation	54.0 days (45-58; n=94)	52.6 days (46-66; n=148)	52.8 days (45-68; n=158)	51.0 days (44-62; n=133)	54.6 days (48-68; n=184)	52.4 days (45-68; n=152)

* Hatching Success = (hatched eggs/total number of eggs) x 100

Table 2. Observed loggerhead turtle bycatch rates in the SEAMAP analysis data set.

YEAR	LOGGERHEADS	LOGGERHEADS PER TOW	LOGGERHEADS PER STANDARD UNIT OF EFFORT*
1990	8	0.0261	0.0894
1991	8	0.0258	0.0894
1992	9	0.0288	0.1006
1993	6	0.0192	0.0671
1994	12	0.0387	0.1342
1995	5	0.0160	0.0559
1996	9	0.0288	0.1006
1997	14	0.0449	0.1565
1998	19	0.0609	0.2124
1999	11	0.0353	0.1230
2000	19	0.0609	0.2124

* The standard unit of effort is a one hour tow with a 100 foot headrope.

Table 3. Results of the stepwise procedure to develop the proportion positive bycatch rate model for the SEAMAP analysis data set.

FACTOR	df	deviance	Deviance/df	% diff.	delta%	L	ChiSquare	Pr>Chi
NULL	3421	999.7656	0.2922			-499.88.	.	
LAT	3420	969.0417	0.2833	3.046	3.046	-484.52	30.7239	0
YEAR	3411	982.8878	0.2882	1.369		-491.44	16.8778	0.07711
PRECIP	3420	990.6407	0.2897	0.856		-495.32	9.1249	0.00252
SURFACE_SALINITY	3416	991.6322	0.2903	0.650		-495.82	1.1198	0.28997
SURFACE_TEMP	3416	992.4236	0.2905	0.582		-496.21	0.3283	0.56665
NORTWIND	3366	978.7567	0.2908	0.479		-489.38	3.8778	0.04893
BOTTOM_SALINITY	3406	991.6663	0.2912	0.342		-495.83	0.4123	0.52082
BOTTOM_TEMP	3406	991.9716	0.2912	0.342		-495.99	0.1069	0.74366
WESTWIND	3366	982.3208	0.2918	0.137		-491.16	0.3136	0.57546
SEASON	3420	999.4015	0.2922	0.000		-499.7	0.3641	0.54624
START_TIME	3420	999.4426	0.2922	0.000		-499.72	0.323	0.56982
START_DEPTH	3420	999.4965	0.2923	-0.034		-499.75	0.2691	0.60393
BAROMETRIC	3420	999.7322	0.2923	-0.034		-499.87	0.0334	0.85489
AIR_TEMP	3418	999.4576	0.2924	-0.068		-499.73	0.1724	0.67795
LAT+								
YEAR	3410	951.9734	0.2792	4.449	1.403	-475.99	17.0683	0.07287
PRECIP	3419	960.4634	0.2809	3.867		-480.23	8.5783	0.0034
SURFACE_TEMP	3415	961.4029	0.2815	3.662		-480.7	1.4663	0.22594
SURFACE_SALINITY	3415	962.3245	0.2818	3.559		-481.16	0.5447	0.46049
BOTTOM_TEMP	3405	961.5905	0.2824	3.354		-480.8	0.878	0.34875
BOTTOM_SALINITY	3405	962.1786	0.2826	3.285		-481.09	0.2899	0.59031
NORTWIND	3365	951.4821	0.2828	3.217		-475.74	1.868	0.17171
AIR_TEMP	3417	967.0103	0.283	3.149		-483.51	1.8022	0.17945
START_DEPTH	3419	967.7791	0.2831	3.114		-483.89	1.2627	0.26115
WESTWIND	3365	953.2602	0.2833	3.046		-476.63	0.0899	0.76437
SEASON	3419	968.6614	0.2833	3.046		-484.33	0.3803	0.53742
START_TIME	3419	968.94	0.2834	3.012		-484.47	0.1017	0.7498
BAROMETRIC	3419	969.0391	0.2834	3.012		-484.52	0.0026	0.95909
LAT+YEAR+								
PRECIP	3409	942.3413	0.2764	5.407	0.958	-471.17	9.6321	0.00191
SURFACE_TEMP	3405	944.5417	0.2774	5.065		-472.27	2.1093	0.1464
SURFACE_SALINITY	3405	946.3728	0.2779	4.894		-473.19	0.2782	0.5979
BOTTOM_TEMP	3395	944.9606	0.2783	4.757		-472.48	1.3408	0.24689
NORTWIND	3355	934.1122	0.2784	4.723		-467.06	2.1519	0.14239
START_DEPTH	3409	949.6511	0.2786	4.654		-474.83	2.3223	0.12753
BOTTOM_SALINITY	3395	946.081	0.2787	4.620		-473.04	0.2204	0.63877
AIR_TEMP	3407	949.7093	0.2788	4.586		-474.85	2.0874	0.14852
WESTWIND	3355	935.9986	0.279	4.517		-468	0.2655	0.60635
BAROMETRIC	3409	951.2455	0.279	4.517		-475.62	0.7279	0.39358
SEASON	3409	951.5995	0.2791	4.483		-475.8	0.3739	0.54088
START_TIME	3409	951.7685	0.2792	4.449		-475.88	0.2049	0.65082
LAT+YEAR+								
LAT*YEAR	3400	934.7971	0.2749	5.921	0.513	-467.4	17.1763	0.07055

% diff: percent difference in deviance/df between each factor and the null model; delta%: percent difference in deviance/df between the newly included factor and the previous factor entered into the model; L: log likelihood; ChiSquare: Pearson Chi-square statistic; Pr>Chi: significance level of the Chi-square statistic.

FINAL MODEL: LAT + YEAR

Table 4. Results of the stepwise procedure to develop the positive bycatch rate model for the SEAMAP analysis dataset.

FACTOR	df	deviance	Deviance/df	% diff.	delta%	L	ChiSquare	Pr>Chi
NULL	113	4.3251	0.0383			-113.8448.	.	
START_TIME	112	4.2286	0.0378	1.305	1.305	-113.7965	0.0966	0.75599
LAT	112	4.2334	0.0378	1.305		-113.7989	0.0918	0.76195
AIR_TEMP	112	4.2985	0.0384	-0.261		-113.8315	0.0266	0.87034
START_DEPTH	112	4.3002	0.0384	-0.261		-113.8323	0.0249	0.87455
SEASON	112	4.3014	0.0384	-0.261		-113.8329	0.0237	0.87761
BAROMETRIC	112	4.3197	0.0386	-0.783		-113.8421	0.0054	0.9413
NORTWIND	110	4.2522	0.0387	-1.044		-111.8084	0.0675	0.79508
SURFACE_TEMP	111	4.2948	0.0387	-1.044		-112.8296	0.0277	0.8679
BOTTOM_TEMP	111	4.3037	0.0388	-1.305		-112.8341	0.0187	0.89117
BOTTOM_SALINITY	111	4.321	0.0389	-1.567		-112.8428	0.0014	0.97014
SURFACE_SALINITY	111	4.3224	0.0389	-1.567		-112.8434	0.0001	0.99237
WESTWIND	110	4.3144	0.0392	-2.350		-111.8394	0.0053	0.94192
YEAR	103	4.071	0.0395	-3.133		-113.7177	0.2541	1

% diff: percent difference in deviance/df between each factor and the null model; delta%: percent difference in deviance/df between the newly included factor and the previous factor entered into the model; L: log likelihood; ChiSquare: Pearson Chi-square statistic; Pr>Chi: significance level of the Chi-square statistic.

FINAL MODEL: YEAR

NOTE: No factors were found to be significant. Year was included in the final model as this is the factor of interest for which least-square means are calculated.

Table 5. Results of the loggerhead turtle bycatch analysis (1990-2000) in SEAMAP analysis dataset.

Lo method with binomial error assumption for proportion positives.

Class Level Information												
Class	Levels	Values										
year	11	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000

Model Fitting Information for _Z									
Weighted by _W									
Description	Value								
Res Log Likelihood	-11114.2								
Akaike's Information Criterion	-11115.2								
Schwarz's Bayesian Criterion	-11118.3								
-2 Res Log Likelihood	22228.5								

Solution for Fixed Effects									
Effect	YEAR	Estimate	Std Error	DF	t	Pr > t	Alpha		
Intercept		7.1539	1.7880	3410	4.00	<.0001	0.05	3.6482	10.6596
lat		-0.3099	0.05585	3410	-5.55	<.0001	0.05	-0.4194	-0.2004
year	1990	-0.8199	0.4408	3410	-1.86	0.0630	0.05	-1.6842	0.04450
year	1991	-0.8537	0.4409	3410	-1.94	0.0529	0.05	-1.7181	0.01070
year	1992	-0.8571	0.4408	3410	-1.94	0.0520	0.05	-1.7214	0.007256
year	1993	-1.1535	0.4869	3410	-2.37	0.0179	0.05	-2.1082	-0.1988
year	1994	-0.5189	0.3993	3410	-1.30	0.1938	0.05	-1.3017	0.2639
year	1995	-1.3396	0.5209	3410	-2.57	0.0102	0.05	-2.3609	-0.3184
year	1996	-0.7338	0.4244	3410	-1.73	0.0839	0.05	-1.5658	0.09829
year	1997	-0.3485	0.3808	3410	-0.92	0.3602	0.05	-1.0952	0.3982
year	1998	-0.06168	0.3556	3410	-0.17	0.8623	0.05	-0.7590	0.6356
year	1999	-0.5236	0.3992	3410	-1.31	0.1898	0.05	-1.3063	0.2592
year	2000	0

Type 3 Tests of Fixed Effects						
Effect	Num DF	Den DF	Chi-Square	F Value	Pr > ChiSq	Pr > F
lat	1	3410	30.79	30.79	<.0001	<.0001
year	10	3410	16.22	1.62	0.0936	0.0942

Least Squares Means										
Effect	year	Margins	Estimate	Error	DF	t Value	Pr > t	Alpha	Lower	Upper
year	1990	WORK._DS	-3.7294	0.3669	3410	-10.17	<.0001	0.05	-4.4487	-3.0101
year	1991	WORK._DS	-3.7633	0.3675	3410	-10.24	<.0001	0.05	-4.4838	-3.0428
year	1992	WORK._DS	-3.7667	0.3674	3410	-10.25	<.0001	0.05	-4.4870	-3.0463
year	1993	WORK._DS	-4.0631	0.4217	3410	-9.64	<.0001	0.05	-4.8898	-3.2363
year	1994	WORK._DS	-3.4285	0.3161	3410	-10.84	<.0001	0.05	-4.0483	-2.8086
year	1995	WORK._DS	-4.2492	0.4605	3410	-9.23	<.0001	0.05	-5.1520	-3.3463
year	1996	WORK._DS	-3.6433	0.3474	3410	-10.49	<.0001	0.05	-4.3245	-2.9622
year	1997	WORK._DS	-3.2581	0.2924	3410	-11.14	<.0001	0.05	-3.8313	-2.6848
year	1998	WORK._DS	-2.9713	0.2584	3410	-11.50	<.0001	0.05	-3.4779	-2.4647
year	1999	WORK._DS	-3.4331	0.3161	3410	-10.86	<.0001	0.05	-4.0529	-2.8134
year	2000	WORK._DS	-2.9096	0.2518	3410	-11.55	<.0001	0.05	-3.4033	-2.4158

Table 6. Results of the loggerhead turtle bycatch analysis (1990-2000) in SEAMAP analysis data set.

Lo method with binomial error assumption for positive bycatch tows.

Class Level Information												
Class		Levels	Values									
year	11	1990	1991	1992	1993	1994	1995	1996	1997	1998	1999	2000
Model Fitting Information for _Z												
Weighted by _W												
Description		Value										
Res Log Likelihood		0.4										
Akaike's Information Criterion		-0.6										
Schwarz's Bayesian Criterion		-1.9										
-2 Res Log Likelihood		-0.9										

Effect	year	Estimate	Standard			t Value	Pr > t	Alpha	Lower	Upper
			Error	DF	DF					
Intercept		0.05407	0.05025	103	1.08	0.2845	0.05	-0.04560	0.1537	
year	1990	-0.05407	0.09232	103	-0.59	0.5594	0.05	-0.2372	0.1290	
year	1991	-0.05407	0.09232	103	-0.59	0.5594	0.05	-0.2372	0.1290	
year	1992	0.06372	0.08864	103	0.72	0.4739	0.05	-0.1121	0.2395	
year	1993	-0.05407	0.1026	103	-0.53	0.5993	0.05	-0.2575	0.1494	
year	1994	0.03294	0.08077	103	0.41	0.6842	0.05	-0.1272	0.1931	
year	1995	-0.05407	0.1101	103	-0.49	0.6244	0.05	-0.2724	0.1643	
year	1996	-0.05407	0.08864	103	-0.61	0.5432	0.05	-0.2299	0.1217	
year	1997	0.02004	0.07715	103	0.26	0.7956	0.05	-0.1330	0.1731	
year	1998	0.05716	0.07107	103	0.80	0.4231	0.05	-0.08379	0.1981	
year	1999	-0.05407	0.08299	103	-0.65	0.5162	0.05	-0.2187	0.1105	
year	2000	0	

Type 3 Tests of Fixed Effects						
Effect	Num	Den	Chi-Square	F Value	Pr > ChiSq	Pr > F
year	10	103	5.30	0.53	0.8703	0.8654

Least Squares Means											
Effect	year	Margins	Estimate	Standard			t Value	Pr > t	Alpha	Lower	Upper
				Error	DF	DF					
year	1990	WORK._DS	1.39E-17	0.07744	103	0.00	1.0000	0.05	-0.1536	0.1536	
year	1991	WORK._DS	1.39E-17	0.07744	103	0.00	1.0000	0.05	-0.1536	0.1536	
year	1992	WORK._DS	0.1178	0.07301	103	1.61	0.1098	0.05	-0.02702	0.2626	
year	1993	WORK._DS	3.47E-17	0.08942	103	0.00	1.0000	0.05	-0.1774	0.1774	
year	1994	WORK._DS	0.08701	0.06323	103	1.38	0.1718	0.05	-0.03840	0.2124	
year	1995	WORK._DS	1.39E-17	0.09796	103	0.00	1.0000	0.05	-0.1943	0.1943	
year	1996	WORK._DS	1.39E-17	0.07301	103	0.00	1.0000	0.05	-0.1448	0.1448	
year	1997	WORK._DS	0.07411	0.05854	103	1.27	0.2084	0.05	-0.04200	0.1902	
year	1998	WORK._DS	0.1112	0.05025	103	2.21	0.0291	0.05	0.01156	0.2109	
year	1999	WORK._DS	2.78E-17	0.06604	103	0.00	1.0000	0.05	-0.1310	0.1310	
year	2000	WORK._DS	0.05407	0.05025	103	1.08	0.2845	0.05	-0.04560	0.1537	

Table 7. Loggerhead turtle relative abundance indices in SEAMAP analysis data set.

YEAR	value	c.v
1990	0.753	0.734
1991	0.728	0.744
1992	0.817	0.697
1993	0.543	0.963
1994	1.101	0.539
1995	0.452	1.137
1996	0.819	0.671
1997	1.281	0.473
1998	1.75	0.375
1999	1.004	0.565

Table 8. Published von Bertalanffy growth curves based on mark-recapture studies of loggerhead sea turtles from the Southeast U.S.

<i>Source</i>	<i>Parameters</i>	<i>N</i>	<i>Study Region</i>	<i>Size Range of Turtles in Study (Initial Capture)</i>	<i>Time Interval Between Captures</i>
Braun-McNeill (in prep ⁵¹)	a=106.9 k=0.0521	57	NC	45.1-75.8 cm SCL	0.936-3.523 yrs.
Foster (1994)	a=96.74 k=0.0637	54	Southeast US	62.2-104.2 cm SCL	1-2186 days
Frazer (1987)	a=94.7 k=0.115	41	FL	N=8: 53.3-77.3 cm SCL N=20: Adults, lengths not specified. N=13: Not specified.	N=8: 0.25-1.64 yrs. N=20: 1.0-4.1 yrs N=13: Not specified.
Henwood (1987)	a=110.0 k=0.0313	118	FL, GA, SC	45-110 cm SCL, t-t total for study (N=3679). Not specified for N=118.	> 90 days
Schmid (1995)*	a=96.08 k=0.0586	51	FL	38.2-110 cm SCL	Less than 90 days to greater than 365 days.
Schmid (1995)**	a=96.10 k=0.0573	19	FL	38.2-110 cm SCL total for study (N=49), but not specified for N=19.	>365 days

*Compiled from all data in study

**Compiled from occasions where the interval between capture and recapture was greater than 1 year.

⁵¹Braun-McNeill, J., S.P Epperly, L. Avens, and S. Sadove. A preliminary analysis of growth rates of juvenile loggerhead (*Caretta caretta*) sea turtles from North Carolina, U.S.A. Manuscript in preparation.

Table 9. Juvenile loggerheads (<86 cm CCL) that dead stranded between 1995 and 1999 and for which sex was determined via direct examination of the gonads. **A.** Total counts of each sex by zone with sex ratios by region. **B.** Sex ratios by state.

State (zones)	% Female
TX (18-21)	0.742105
FL (1-10, 24-30)	0.655172
GA (30, 31)	0.629464
SC (32, 33)	0.674419
NC (33-36)	0.652542
VA (36-38)	0.674419

A.

Zone	Female	Males	% Female
1	1	0	0
2	0	0	0
3	1	0	0
4	0	0	0
5	2	0	0
6	0	0	0
7	0	0	0
8	2	2	2
9	1	0	0
10	0	0	0
11	0	0	0
12	1	0	0
13	0	0	0
14	0	0	0
15	0	0	0
16	0	0	0
17	0	0	0
18	21	10	
19	8	2	
20	97	30	
21	15	7	
22	0	0	
23	0	0	
Gulf:	149	51	0.745
24	1	1	
25	7	4	
26	2	3	
27	11	1	
28	10	4	
SE FL:	31	13	0.705
29	12	6	
30	102	59	
31	39	24	
32	20	10	
33	9	4	
34	28	18	
35	28	13	
36	12	6	
37	12	6	
38	5	2	
39	2	3	
40	16	8	
41	62	8	
42	0	0	
43	0	0	
44	0	0	
NEFL-ME:	347	167	0.675
Total:	527	231	0.695

B.

Stage	Duration	Annual Survival Rate
Pelagic Juvenile	6	Varies
Small Benthic Juvenile	7	0.6758
Large Benthic Juvenile	7	0.7425
Breeding Adult	Indefinite	0.809
Non-breeding Adult	Indefinite	0.809

Stage	Duration	Annual Survival Rate
Pelagic Juvenile	7	Varies
Small Benthic Juvenile	6	0.6758
Large Benthic Juvenile	14	0.7425
Breeding Adult	Indefinite	0.809
Non-breeding Adult	Indefinite	0.809

Stage	Duration	Annual Survival Rate
Pelagic Juvenile	6	Varies
Small Benthic Juvenile	13	0.893
Large Benthic Juvenile	11	0.893
Breeding Adult	Indefinite	0.812
Non-breeding Adult	Indefinite	0.812

Stage	Duration	Annual Survival Rate
Pelagic Juvenile	7	Varies
Small Benthic Juvenile	11	0.893
Large Benthic Juvenile	21	0.893
Breeding Adult	Indefinite	0.812
Non-breeding Adult	Indefinite	0.812

Table 14. Annual pelagic stage survival rates estimated from the 4 model scenarios at 3 values of λ .

	Annual Survival Rate for Pelagic Juveniles			
1	Model 1	Model 2	Model 3	Model 4
0.95	0.744	0.910	0.510	0.585
0.97	0.803	0.990	0.565	0.657
1.0	0.894	>1.000	0.660	0.780

Table 15. Loggerhead turtle strandings by zone, 1998 – 2000.¹² Data for 2000 are preliminary. Cold-stunned turtles, captive-reared turtles and post-hatchlings are not included.

Zone	1998	1999	2000
1	17	19	44
2	5	0	3
3	6	6	19
4	37	48	110
5	39	34	73
6	2	2	3
7	9	6	9
8	22	26	33
9	8	6	16
10	10	9	11
11	19	15	4
12	5	6	1
13	0	0	1
14	5	0	4
15	0	0	0
16	0	0	0
17	8	16	0
18	32	52	37
19	24	40	21
20	65	90	77
21	48	28	27
24	11	14	27
25	34	30	25
26	41	29	54
27	58	50	60
28	102	66	73
29	74	91	58
30	151	128	82
31	127	133	70
32	145	79	81
33	61	58	79
34	87	75	89
35	77	187	396
36	181	164	178
37	100	77	119
38	49	54	38
39	27	48	43
40	24	13	12
41	3	7	12
42	0	1	0

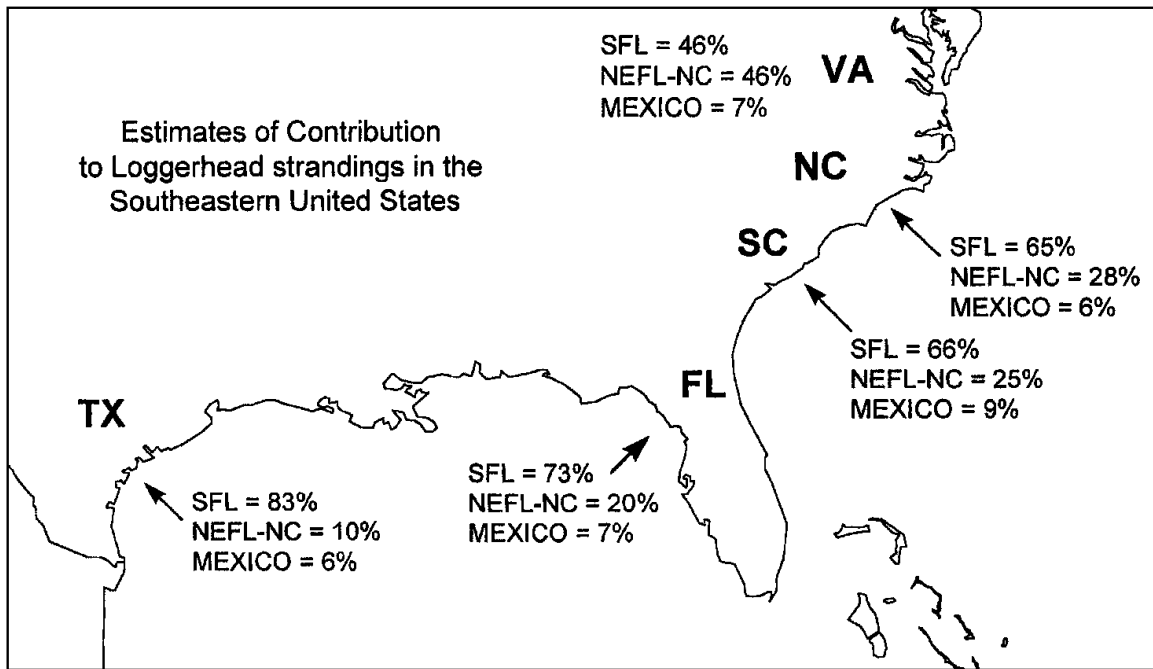


Figure 1. Geographic representation of maximum likelihood estimates of percent contribution to loggerhead strandings in the Southeastern United States. Abbreviations: SFL=South Florida, NEFL-NC=Northeast Florida to North Carolina. Figure is reproduced from Bass *et al.* (1999¹³).

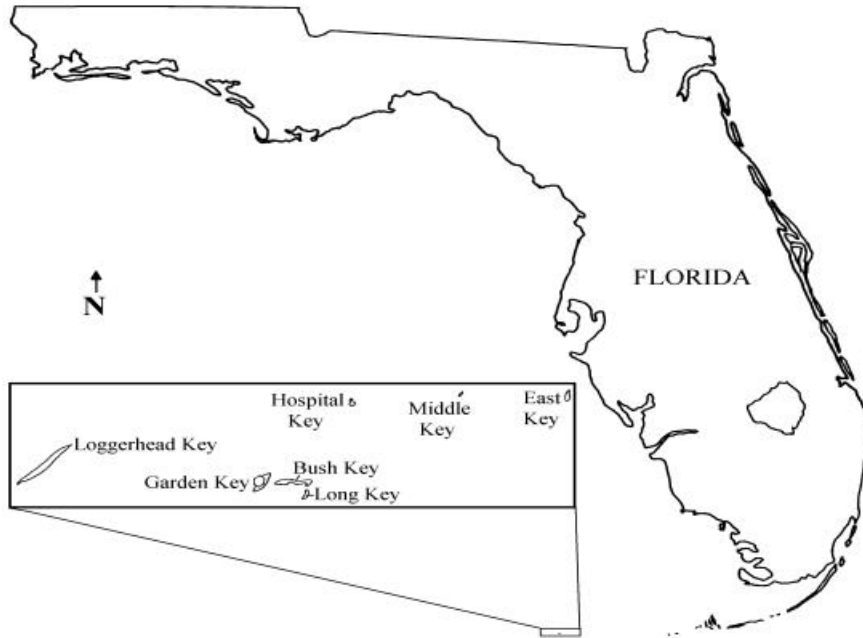


Figure 2. Location of the Dry Tortugas, where loggerhead turtles nest.

Nesters Size Distribution

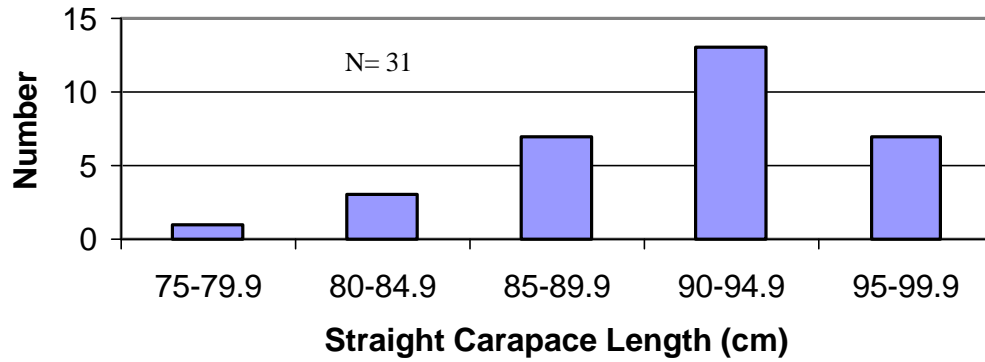


Figure 3 Size distribution of loggerhead turtles nesting in the Dry Tortugas National Park, 1981-1984. The mean straight carapace length was 90.4 cm (CMTTP³).

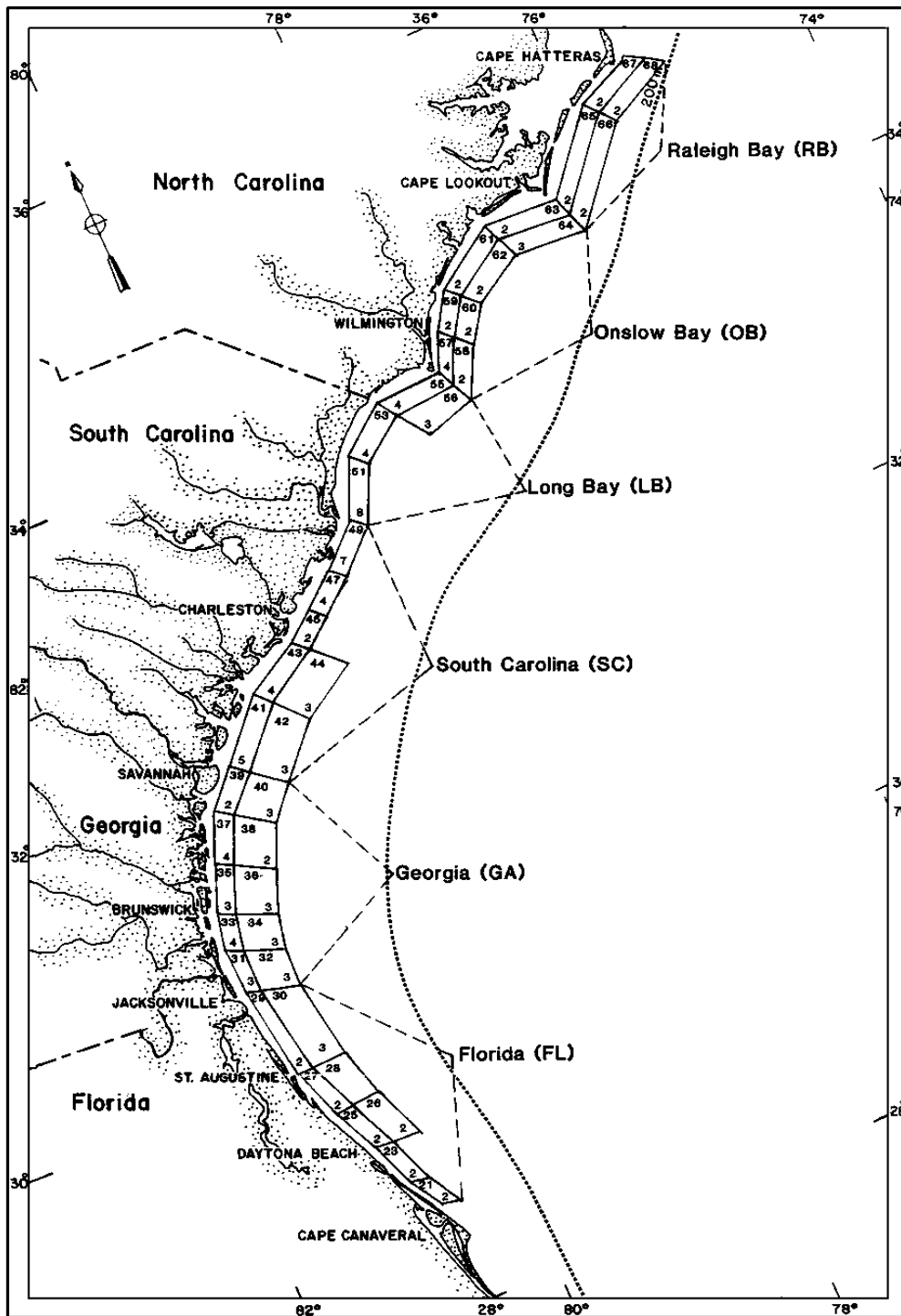


Figure 4. Geographic range of SEAMAP sampling in the Southeast United States. Stratum number is located in the upper left and number of trawl samples collected in the lower right of each stratum. Strata are not drawn to scale. Reprinted from SCMRD (2000).

Loggerhead Observed Bycatch

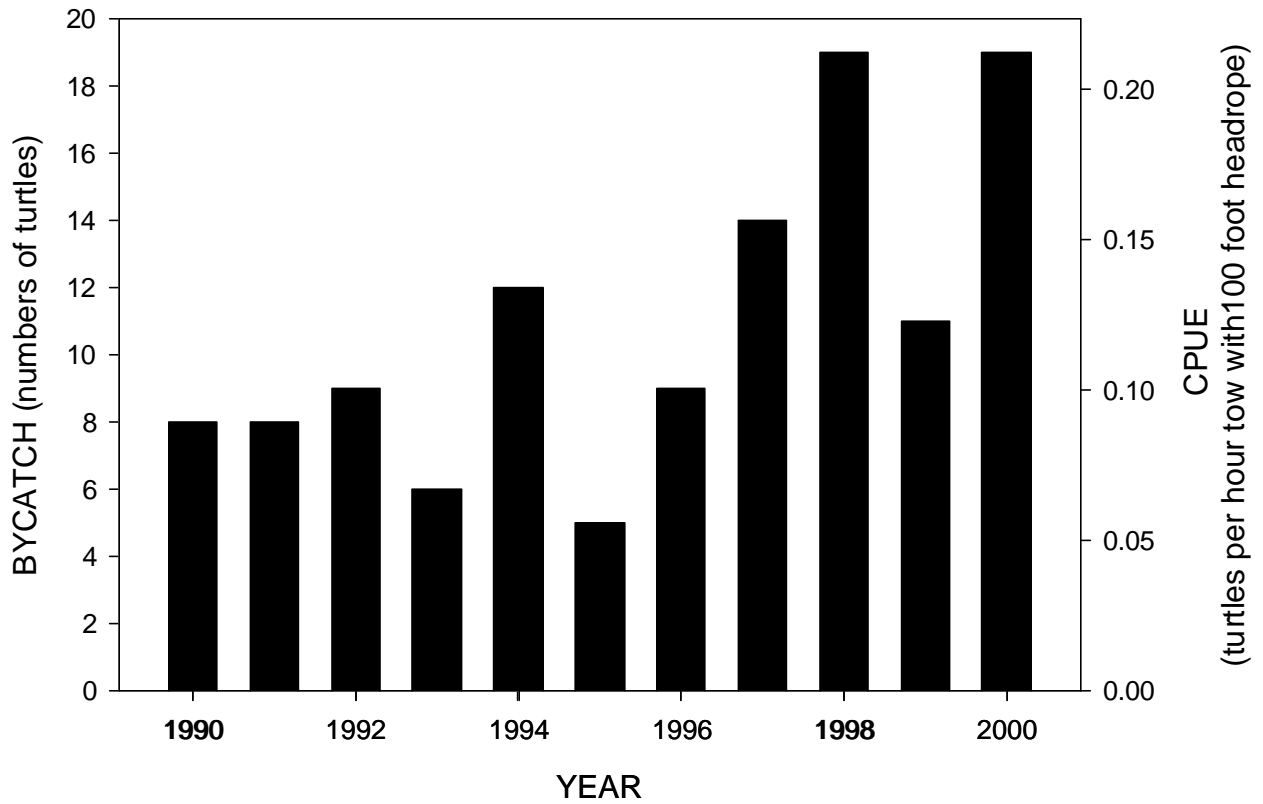


Figure 5. Observed loggerhead turtle bycatch rates in the SEAMAP analysis data set.

Loggerhead Turtles

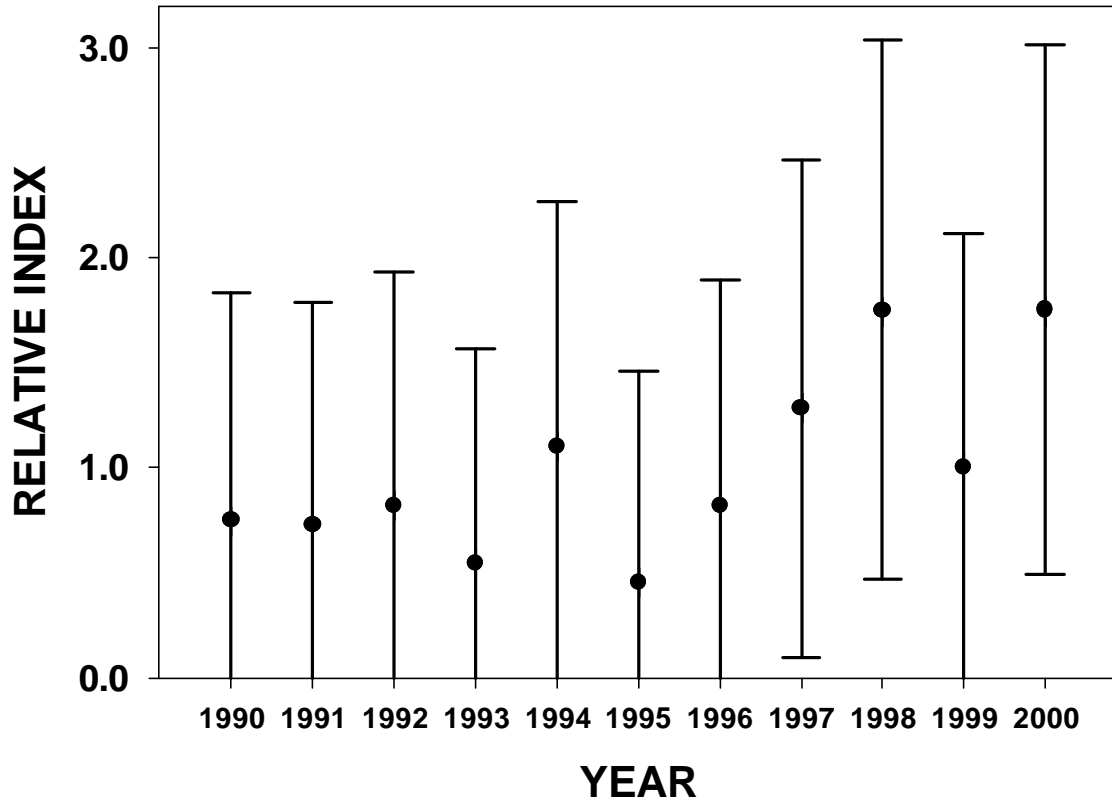


Figure 6. Relative standardized abundance indices for loggerhead turtles in SEAMAP analysis data set with approximate 95% confidence intervals (solid circles) and observed relative bycatch rates (open diamonds).

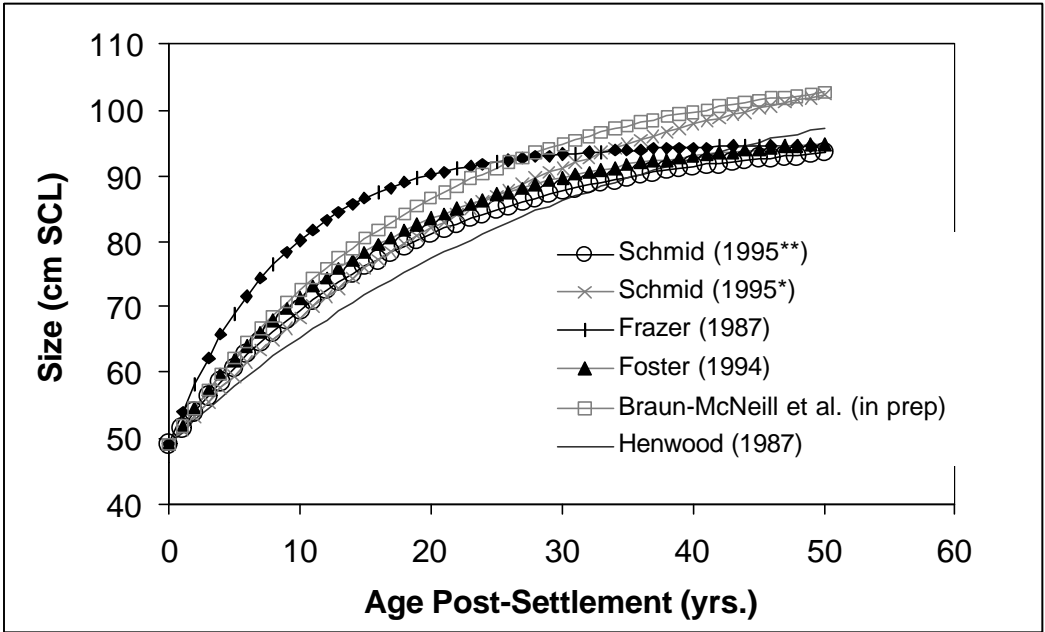


Figure 7. Published von Bertalanffy growth curves (see Table 1 for parameters). Curves were plotted using the equation $y=a-(a-\text{initial size})e^{-kx}$. As only post-settlement growth rates are being considered, 49 cm SCL was used as initial size.

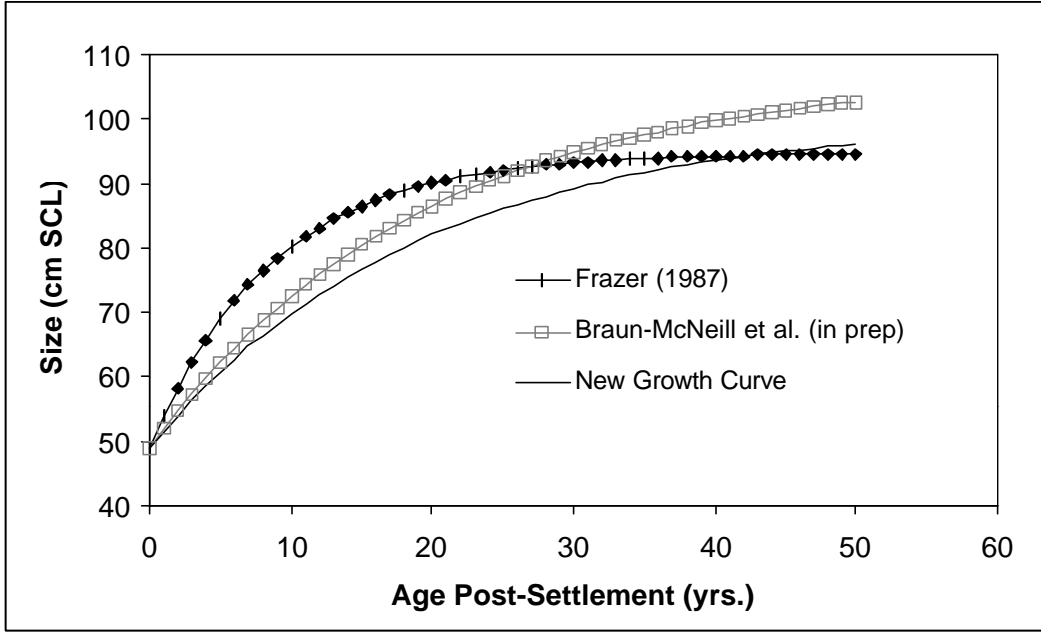


Figure 8. New growth curve generated by adding additional recaptures of turtles > 70 cm SCL to the data from Braun-McNeill et al (2001). The parameters are $a=99.7$ and $k=0.053$. The curve is shown with curves from Frazer (1987) and Braun-McNeill et al (in prep)⁵² for comparison.

⁵² Braun-McNeill, J., S.P Epperly, L. Avens, and S. Sadove. A Preliminary analysis of growth rates of juvenile loggerhead (*Caretta caretta*) sea turtles from North Carolina, U.S.A. Manuscript in preparation.

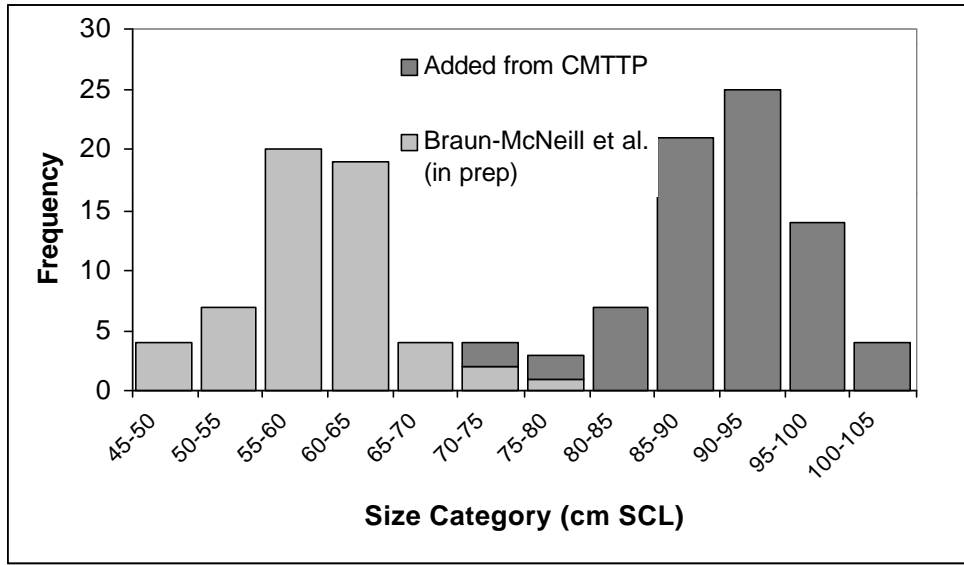


Figure 9. Size distribution of turtles from mark-recapture studies used to estimate a new von Bertalanffy growth curve for loggerhead sea turtles from the Southeast U.S.

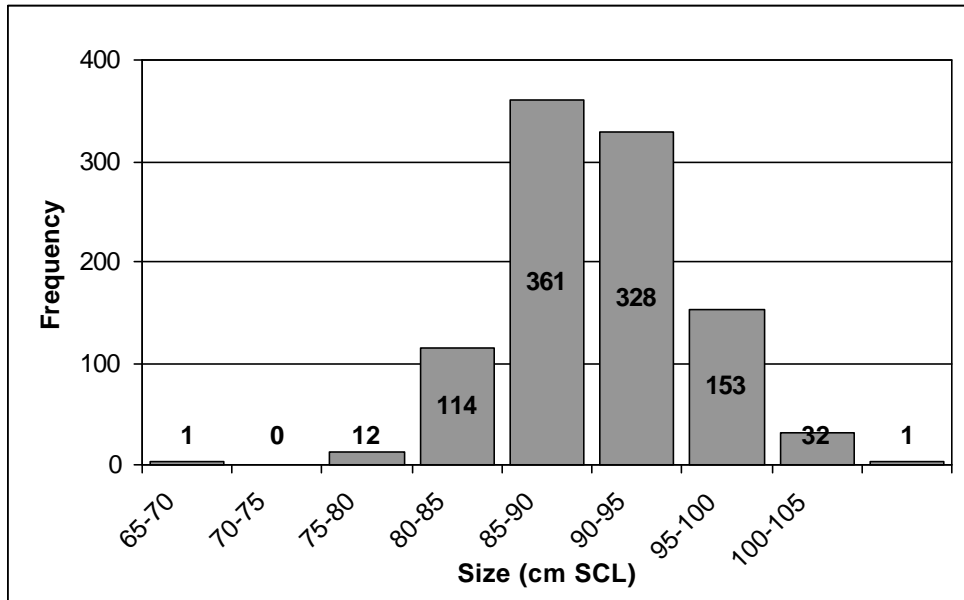


Figure 10. Size frequency of nesting loggerheads from the CMTTP database, using only reported SCL's (no conversions from CCL) and initial captures (no recaptures). Average size is 90.38 cm SCL (SD=5.08). The smallest nester is 68.5 cm SCL and the largest is 105.1 cm SCL.

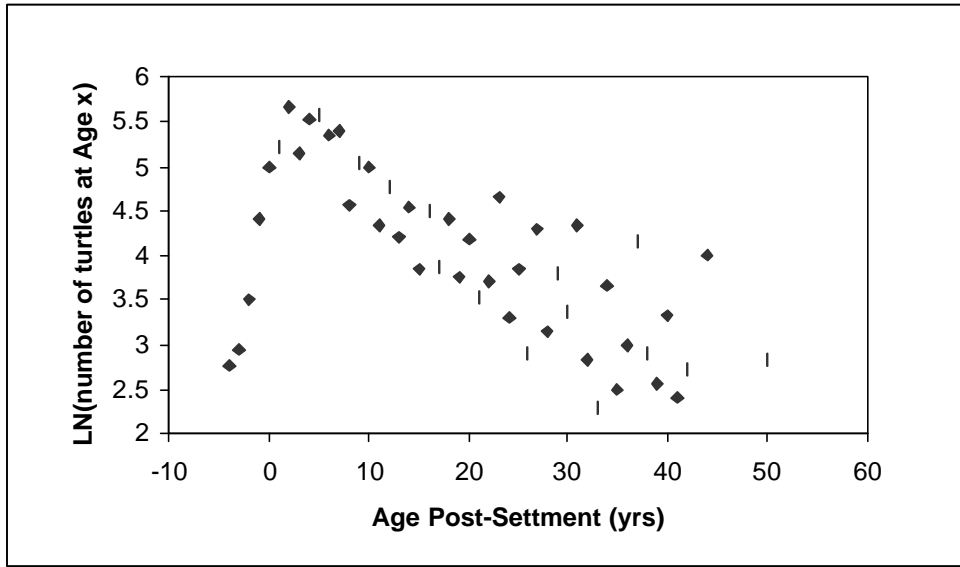


Figure 11. Catch curve for 1986-1989 loggerheads sea turtle strandings, zones 1-35. Size-at-age estimated using the 'New' von Bertalanffy growth curve (see text).

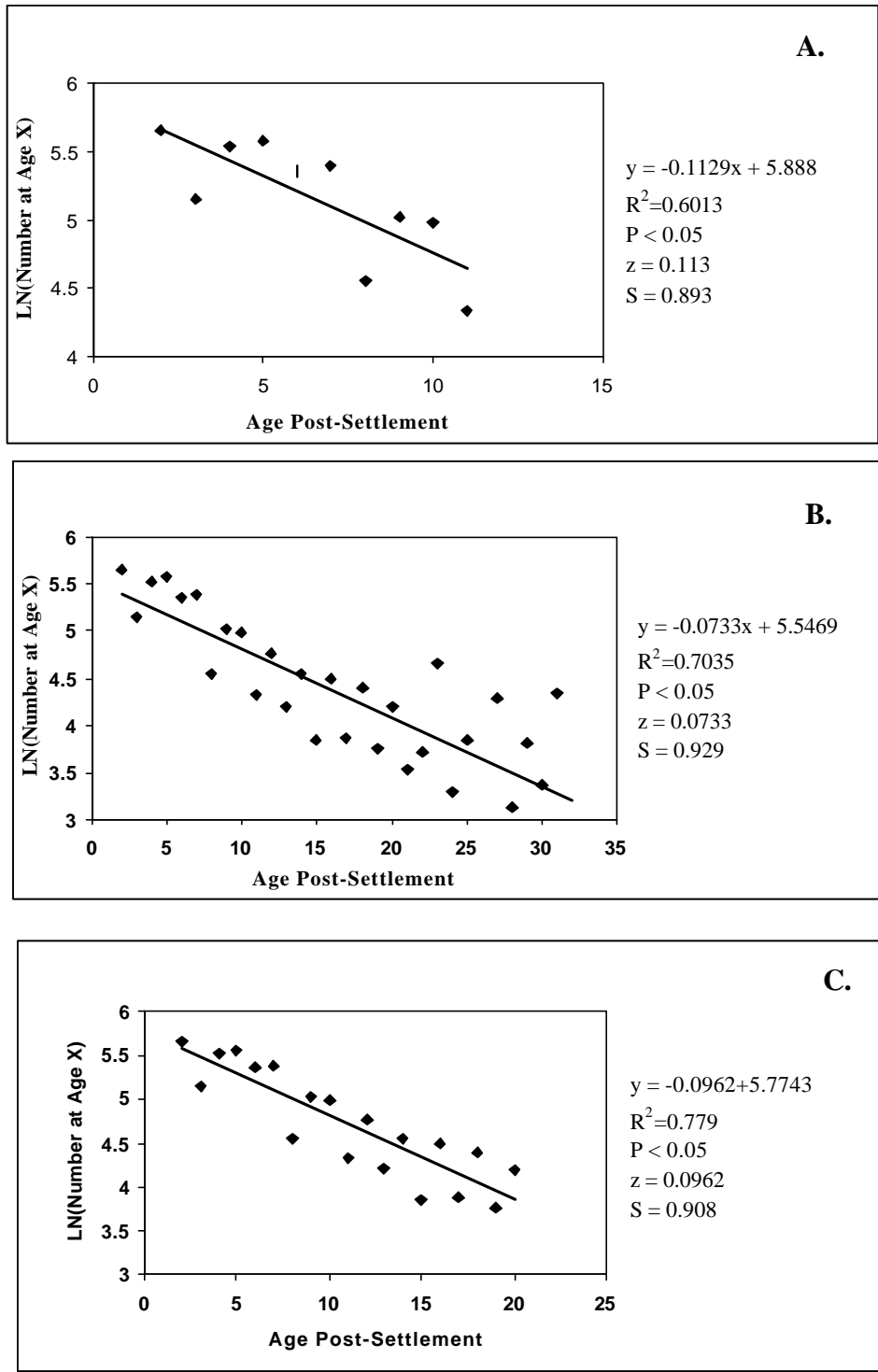


Figure 12. Catch curve from Fig 11 with instantaneous mortality rates (z) and annual survival rate (S) estimated by examining the slope of the declining arm of the catch curve at 3 different points. A) At the age corresponding to 70 cm SCL. B) At the age corresponding to 90 cm SCL. C) At the point where the data begin to scatter (Fig. 5).

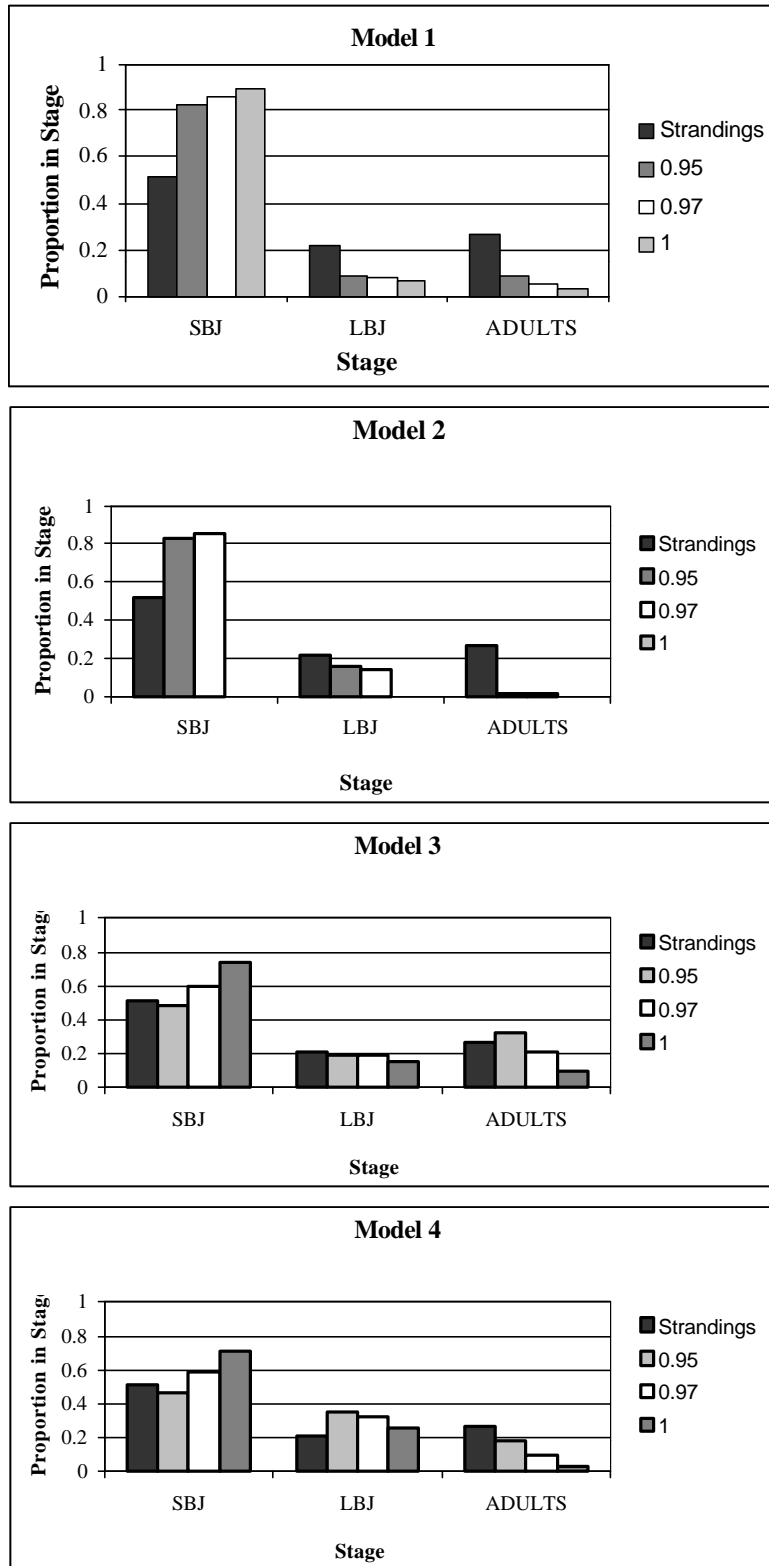


Figure 13. The proportion of animals in the three benthic stages, small benthic juvenile (SBL), large benthic juvenile (LBJ) and adults, predicted by the stable-age distribution of the 4 models. These results are compared to the proportion of animals within each stage based on size from dead strandings in the southeast U.S. from 1986 to 1989.

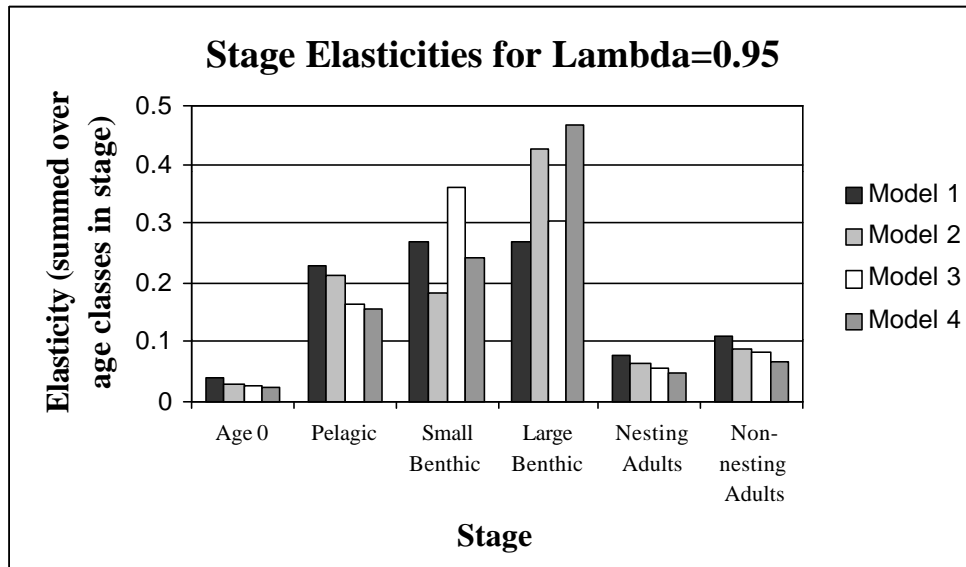


Figure 14. Elasticities summed over all ages in stage. Values given are for proportion female offspring = 0.35 and $\lambda = 0.95$.

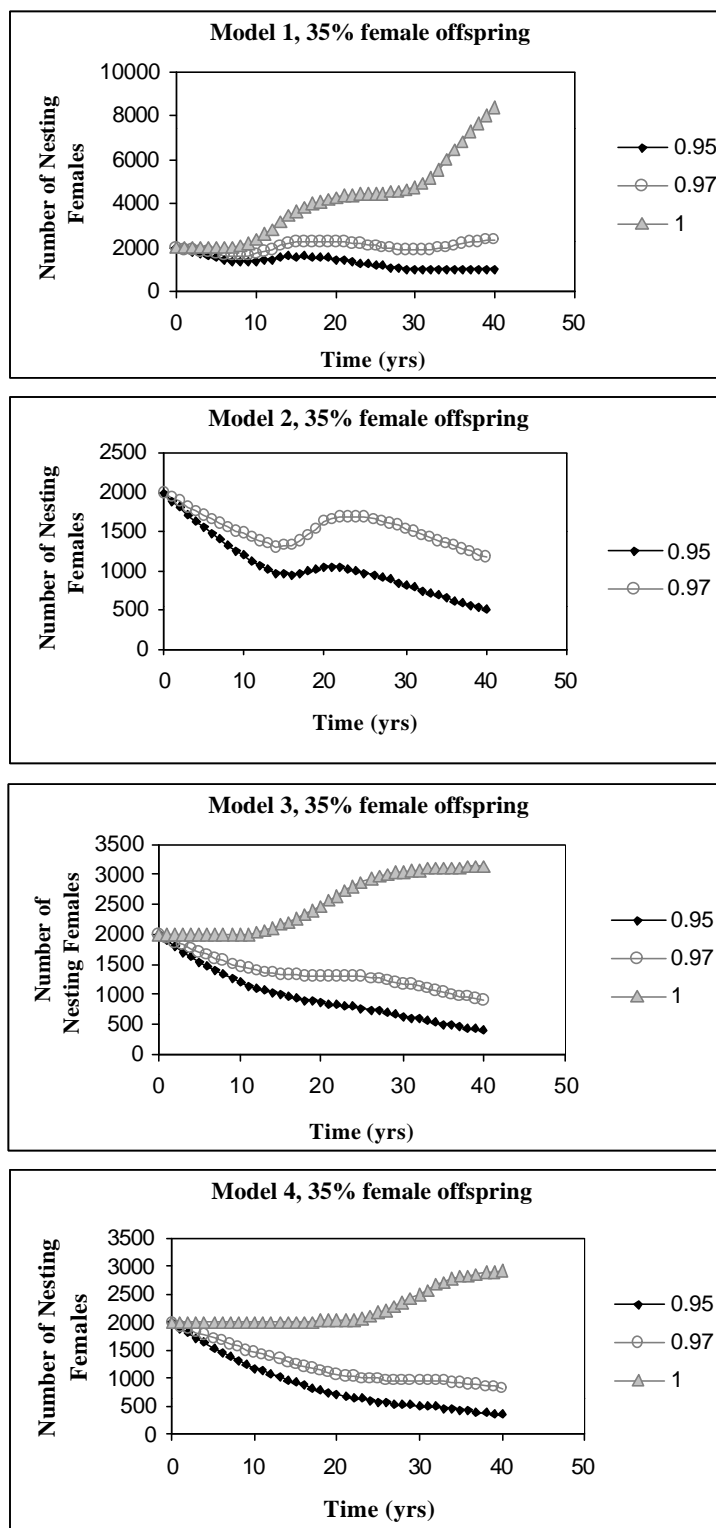


Figure 15. Population trajectories for the 4 models. Model runs were initialized with a population at stable age distribution for the appropriate combination of model and λ , assuming 2000 nesting females. Small benthic juvenile mortality was decreased by 30% and the population projected based on the new survival rates.

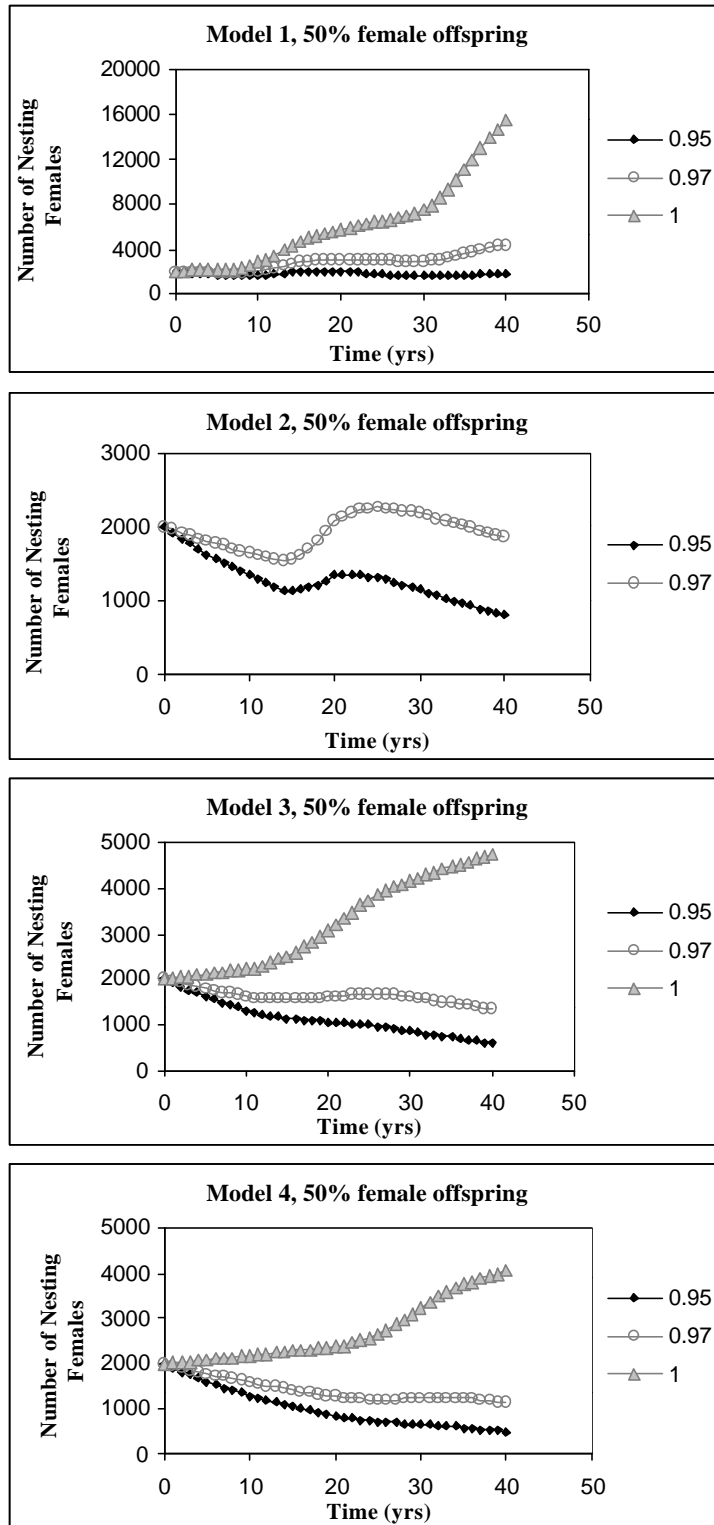


Figure 16. Population trajectories for the same 4 models as in Fig. 14 but with the proportion of female offspring now set to 0.50 in the fecundity function. Model runs were initialized with a population at stable age distribution for the appropriate combination of model and λ , assuming 2000 nesting females. Small benthic juvenile mortality was decreased by 30% and the population projected based on the new survival rates.

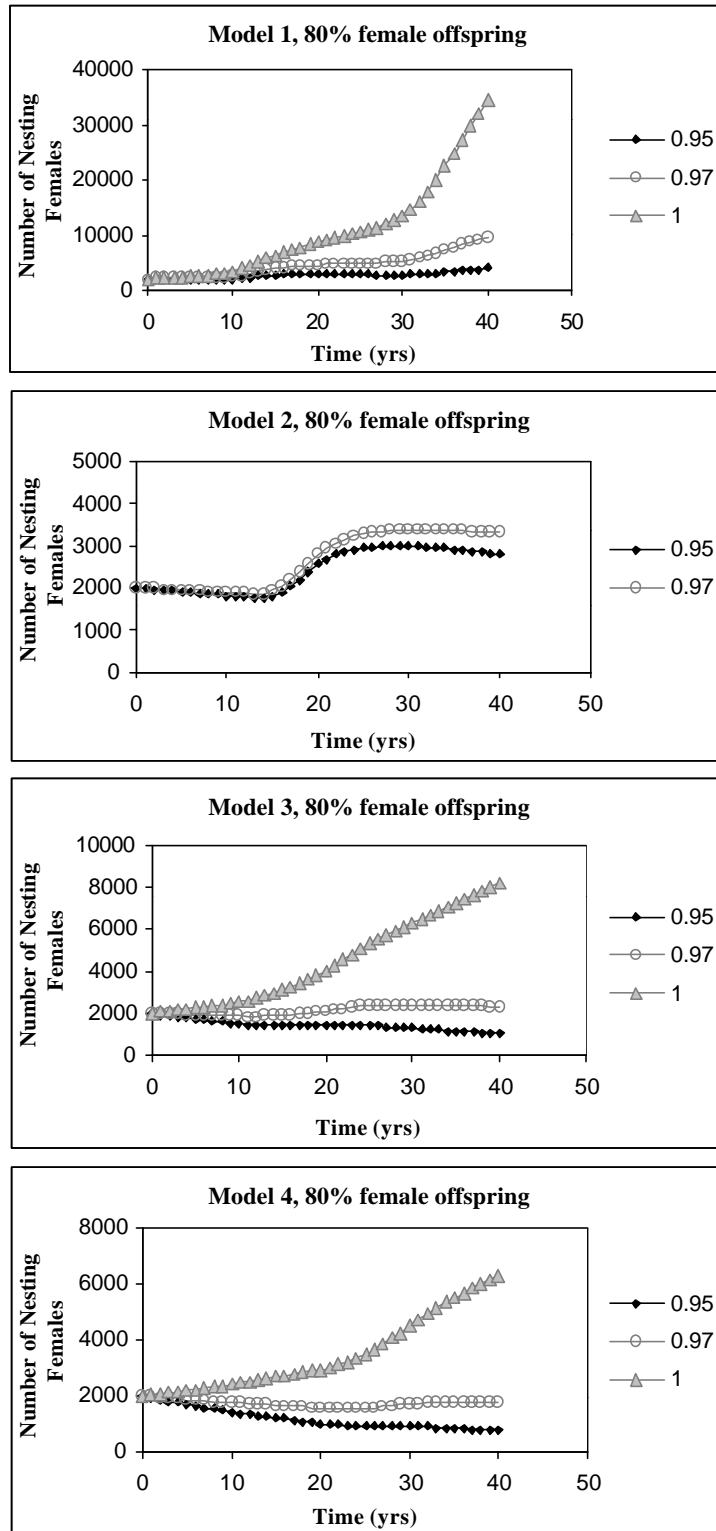


Figure 17. Population trajectories for the same 4 models as in Fig. 14 but with the proportion of female offspring now set to 0.80 in the fecundity function. Model runs were initialized with a population at stable age distribution for the appropriate combination of model and λ , assuming 2000 nesting females. Small benthic juvenile mortality was decreased by 30% and the population projected based on the new survival rates.

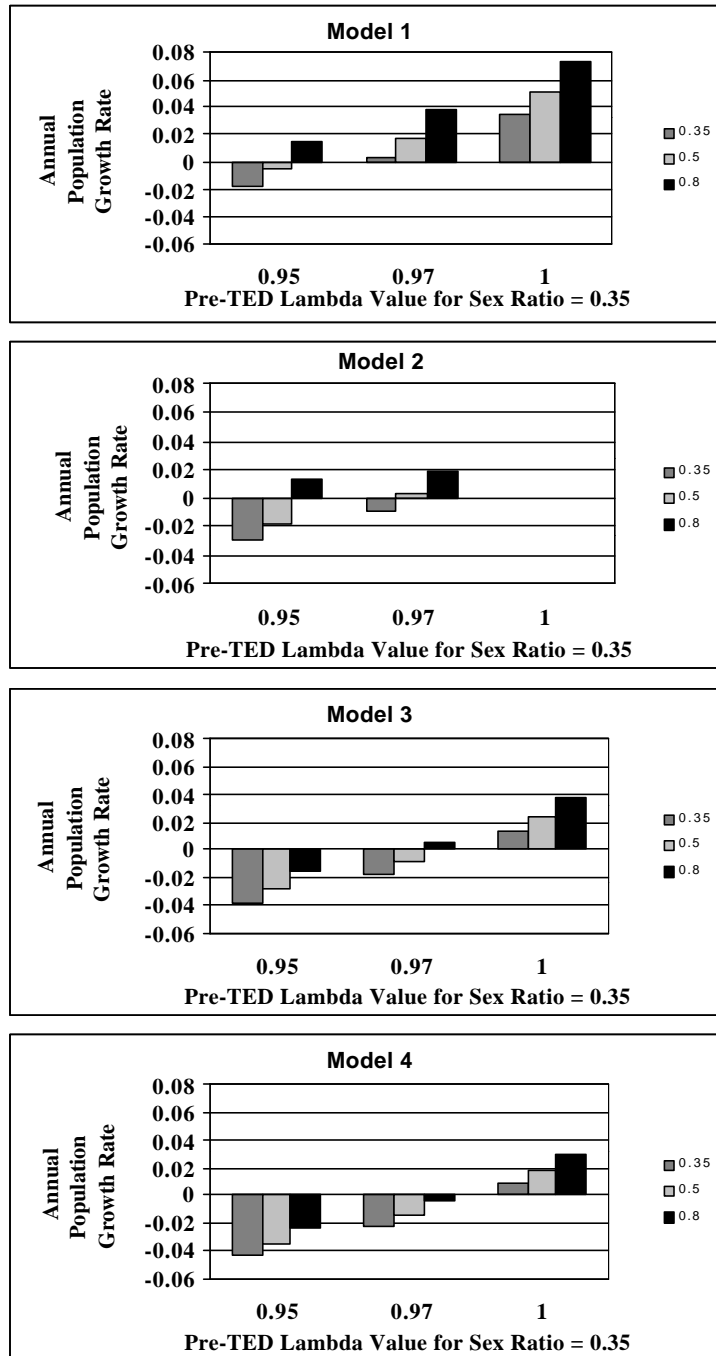


Figure 18. Population growth rates following a 30% reduction in mortality in the small benthic stage. Each model (1-4) was run at 3 initial values of λ (equal to 0.95, 0.97 and 1.0 for proportion of female offspring = 0.35) and at three values for proportion of female offspring (0.35, 0.50 and 0.80).