

**POPULATION SEX RATIO OF THE KEMP'S RIDLEY SEA TURTLE**

***(LEPIDOCHELYS KEMPII):***

**PROBLEMS IN POPULATION MODELING**

A Dissertation

by

MICHAEL SCOTT COYNE

Submitted to the Office of Graduate Studies of  
Texas A&M University  
in partial fulfillment of the requirements for the degree of

DOCTOR OF PHILOSOPHY

May 2000

Major Subject: Wildlife and Fisheries Sciences

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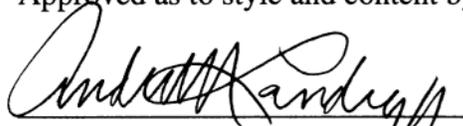
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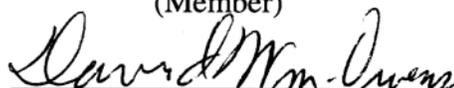
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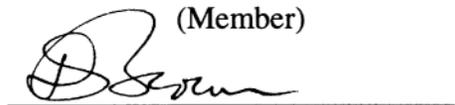
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May 2000

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## ABSTRACT

Population Sex Ratio of the Kemp's Ridley Sea Turtle (*Lepidochelys kempii*):

Problems in Population Modeling. (May 2000)

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Chair of Advisory Committee: Dr. André M. Landry, Jr.

The Kemp's ridley (*Lepidochelys kempii*) is the most endangered sea turtle species in the world and has been the focus of intense recovery efforts in the United States and Mexico. The objectives of this research were to 1) establish testosterone radioimmunoassay sexing criteria for *L. kempii*; 2) estimate a sex ratio for the *L. kempii* population; 3) develop a preliminary population model for *L. kempii*; and 4) assess the impact of various sex ratios on *L. kempii* population dynamics.

This study utilized radioimmunoassay (RIA) determination of blood plasma testosterone (T) concentration in conjunction with limited laparoscopy to sex captured *L. kempii*. These data yielded a plasma T sexing criteria for *L. kempii* of  $\leq 12$  pg/ml for females and  $\geq 18$  pg/ml for males.

The sexing criteria applied to all *L. kempii* for which blood was obtained resulted in 134 females, 95 males and 10 indeterminates (1.5F:1.0M). The capture lot included 20 headstart *L. kempii*, including 2 males and 18 females, resulting in 132 wild females and 99 wild males (1.3F:1.0M).

These results and scientific literature were used to develop a preliminary population model for *L. kempii*. The final model was used to assess the impact of various population sex ratio values upon Kemp's ridley sea turtle demography.

Two scenarios were tested using the model. One in which proportion of adult

males in the population have no effect on hatchling production and another where productivity is a function of the relative abundance of adult males. Under the first scenario, the model suggests that a greater proportion of females in the population dramatically enhances hatchling production, yielding a 271% increase in predicted nests after 50 years with a sex ratio of 3F:1M and an 81% decrease with 1F:3M. The second scenario, in which reproductive output is a function of male availability, was approximated by assuming a linear relationship between increasing proportion of males and reproductive success. Strong female bias (3F:1M) resulted in a 20% decrease in the population while a male bias (1F:3M) yielded a 62% decrease. The greatest rate of reproductive return was achieved with a sex ratio of 1.28F:1M (56.2% female).

**DEDICATION**

to my loving wife  
and Isabel...

## ACKNOWLEDGMENTS

There are many to whom I am indebted for their assistance, advice, and moral support in the completion of my dissertation. Lack of space and a poor memory makes it impossible to name everyone. However, I offer my sincerest gratitude to all who have made this effort possible.

To the members of my committee, Dr. André M. Landry, Jr., Dr. Robert B. Ditton, Dr. David W. Owens, and Dr. Bill Grant I owe many thanks. Their patience and wisdom proved invaluable. Special thanks goes to Dr. André M. Landry, Jr., my principal advisor. His own drive made it possible for me to accomplish my goals and pushed me to always improve myself.

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## CHAPTER I

### INTRODUCTION

The Kemp's ridley (*Lepidochelys kempii*) is the most endangered sea turtle species in the world and, as such, is protected under the Endangered Species Act of 1973 (Public Law 93-205). An estimated 40,000 *L. kempii* nested in a single day in 1947 at Rancho Nuevo, Mexico (Carr, 1963). Exploitation of eggs and adult females on the nesting beach and incidental capture by commercial fisheries reduced the nesting population to a record low 702 nests during 1985, laid by approximately 230 - 350 adult females. Although 3,648 nests (an estimated 1,200 to 1,800 nesting females) were counted at Rancho Nuevo during 1999 (Burchfield *et al.*, 1999), the current *L. kempii* population is a fraction of historic levels. This species' endangered status mandates collection of population structure data and use of population modeling to fill information gaps prerequisite to its management (U.S. Fish and Wildlife Service and National Marine Fisheries Service [USFWS/NMFS], 1992; Marine Turtle Specialist Group [MTSG], 1995).

#### **Kemp's Ridley Life History**

The Kemp's ridley belongs to the Order Chelonia and Family Cheloniidae and is thought to have diverged from the olive ridley (*Lepidochelys olivacea*) as a result of the emergence of the Isthmus of Panama approximately three million years ago (Hendrickson, 1980). Mitochondrial DNA analysis supports this theory, indicating that the olive ridley and Kemp's ridley species diverged during the same era, an estimated 3-6 million years ago (Bowen *et al.*, 1991).

## Distribution

The Kemp's ridley is found in temperate to tropical regions of the western north Atlantic (Fig. 1). Adults are primarily restricted to the Gulf of Mexico (Carr, 1963; Márquez, 1970, 1990; Pritchard and Márquez, 1973; Groombridge, 1982; Wilson and Zug, 1991), while immature specimens occur in the Gulf and, to a lesser extent, along the east coast of North America. Kemp's ridleys have occasionally been reported from the Caribbean; however, Pritchard and Márquez (1973) suggested these observations

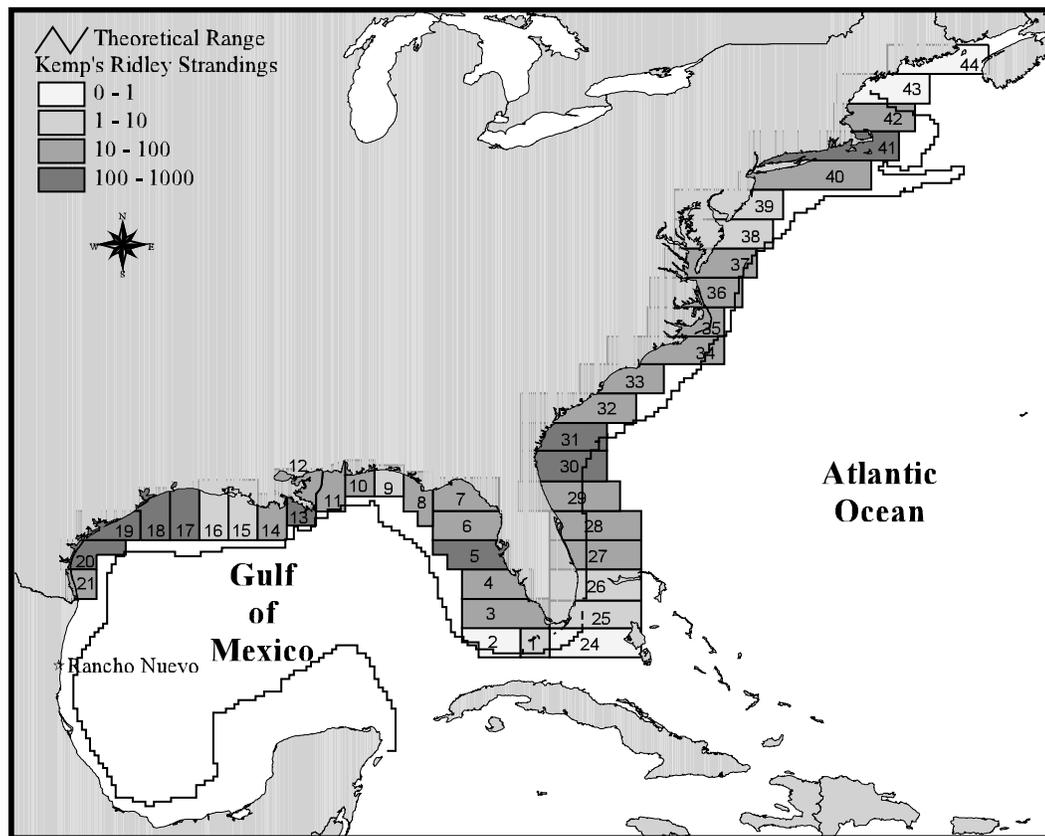


Figure 1. Number of stranded Kemp's ridleys collected in the U.S. by the NMFS Sea Turtle Stranding and Salvage Network during 1986 - 1996 and theoretical distribution (solid line) based on a habitat suitability model (Coyne *et al.*, 2000). Numbers within blocks denote NMFS Statistical Subareas.

may be misidentified olive ridleys (*L. olivacea*).

Kemp's ridleys also have been observed drifting in European waters (Brongersma, 1972). Captive-raised ridleys released off Texas, Florida, and Mexico have been recovered as far away as France, Morocco and Wales (Manzella *et al.*, 1988; Marine Turtle Newsletter, 2000). Carr (1982, 1986) proposed a relationship between seaweed masses and fronts as a pelagic habitat where post-hatchling turtles find food and protection, perhaps facilitating drift to the eastern Atlantic. However, there have been no confirmed reports of hatchling or post-hatchling *L. kempii* in the pelagic zone.

Kemp's ridleys have been the subject of investigations in Florida (Ogren, 1989; Schmid, 1994), Georgia, and South Carolina (Henwood, 1987), as well as, Chesapeake Bay (Byles, 1989), Long Island, New York (Burke *et al.*, 1994), and New England bays (Lazell, 1980). Documented occurrences of *L. kempii* outside the western Gulf of Mexico are most frequent from December to March in southeast Florida waters (Henwood and Ogren, 1987; Ogren, 1989; Rudloe *et al.*, 1991, Schmid, 1994).

Adult female *L. kempii* nest along a narrow beach in the western Gulf of Mexico known as Rancho Nuevo, just south of the Tropic of Cancer in the State of Tamaulipas, Mexico (Fig. 1). A limited number of nests has been reported from other regions including: Veracruz (Carr and Caldwell, 1958), Campeche (Carr *et al.*, 1982; Márquez and Fritts, 1987), Texas (Carr, 1961, 1963; Hildebrand 1963, 1982, 1983; Pritchard and Márquez, 1973; Rabalais and Rabalais, 1980; Carr *et al.*, 1982; Shaver, 1996), Florida (Meylan *et al.*, 1991), South Carolina, North Carolina, and Magdalena, Colombia (Chavez and Kaufman, 1974).

## **Diet**

Neonate *L. kempii* presumably feed on *Sargassum*, associated infauna and other epipelagic prey in the Gulf of Mexico. Post-pelagic ridleys are largely

carnivorous, feeding primarily on blue crabs, *Callinectes sapidus* (Shaver, 1991; Burke *et al.*, 1994; Werner, 1994). The aforementioned studies and other analyses of stomach contents of stranded dead turtles indicate that *L. kempii* is a shallow water, benthic feeder (De Sola and Aabrams, 1933; Carr, 1942; Smith and List, 1950; Liner, 1954; Dobie *et al.*, 1961; Hardy, 1962; Montoya, 1966; Márquez, 1970; Ernst and Barbour, 1972; Pritchard and Márquez, 1973; Hendrickson, 1980; Mortimer, 1981; Hildebrand, 1982; Lutcavage and Musick, 1985). One study of fecal material from at-sea captured turtles (n = 86) suggested that crab (61.57% dry mass), fish (13.65% dry mass) and mollusc (6.55% dry mass) make up the majority of *L. kempii* diet (Werner, 1994).

## **Reproduction**

Principal courtship and mating areas for *L. kempii* are not well known. Anecdotal information supplied by fishermen suggests that mating occurs at or before the nesting season near the nesting beach (Chavez *et al.*, 1967; Pritchard, 1969; Márquez, 1970). Shaver (1991) reported a mating pair of *L. kempii* in Mansfield Channel, Texas at the southern boundary of the Padre Island National Seashore. The bulk of *L. kempii* females reproduce annually from April into July (Márquez, 1982a).

Nesting *L. kempii* females typically come ashore in groups called “arribadas” to dig their nests and lay eggs. Kemp’s ridley, the smallest sea turtle species, digs the shallowest nests (35 - 40 cm), requiring 10 - 15 minutes for excavation (Márquez, 1994). After depositing her eggs, the female quickly covers the nest and returns to sea. Mean clutch size is reported to be 100.8 eggs (USFWS/NMFS, 1992). Estimates of seasonal clutch frequency range from 1.8 (Turtle Expert Working Group [TEWG], 1998) to 3.075 nests/female/season (Rostal, 1991). Adult female *L. kempii* are estimated to nest approximately every 1.89 years (TEWG, 1998). Hatchlings emerge after 45-58 days, the length of incubation depending primarily upon incubation temperature.

## Life History

**Pelagic Post-hatchling:** Ogren (1989) defines pelagic stage post-hatchling *L. kempii* as those individuals less than 20 cm straight carapace length (SCL). Pelagic *L. kempii* hatchlings swim across the narrow continental shelf off Rancho Nuevo and are entrained in a western Gulf of Mexico loop current (Fig. 2; Collard and Ogren, 1990). The loop current may carry post-hatchling Kemp’s ridleys to the northern Gulf, the central-southwestern Gulf, or sweep them through the Straits of Florida and northward in the Florida Current and Gulf Stream. It remains unknown what portion, if any, of the latter is lost to the western Gulf of Mexico reproductive population. However, the recently recorded nesting at Rancho Nuevo of four turtles originally tagged on the U.S. east coast

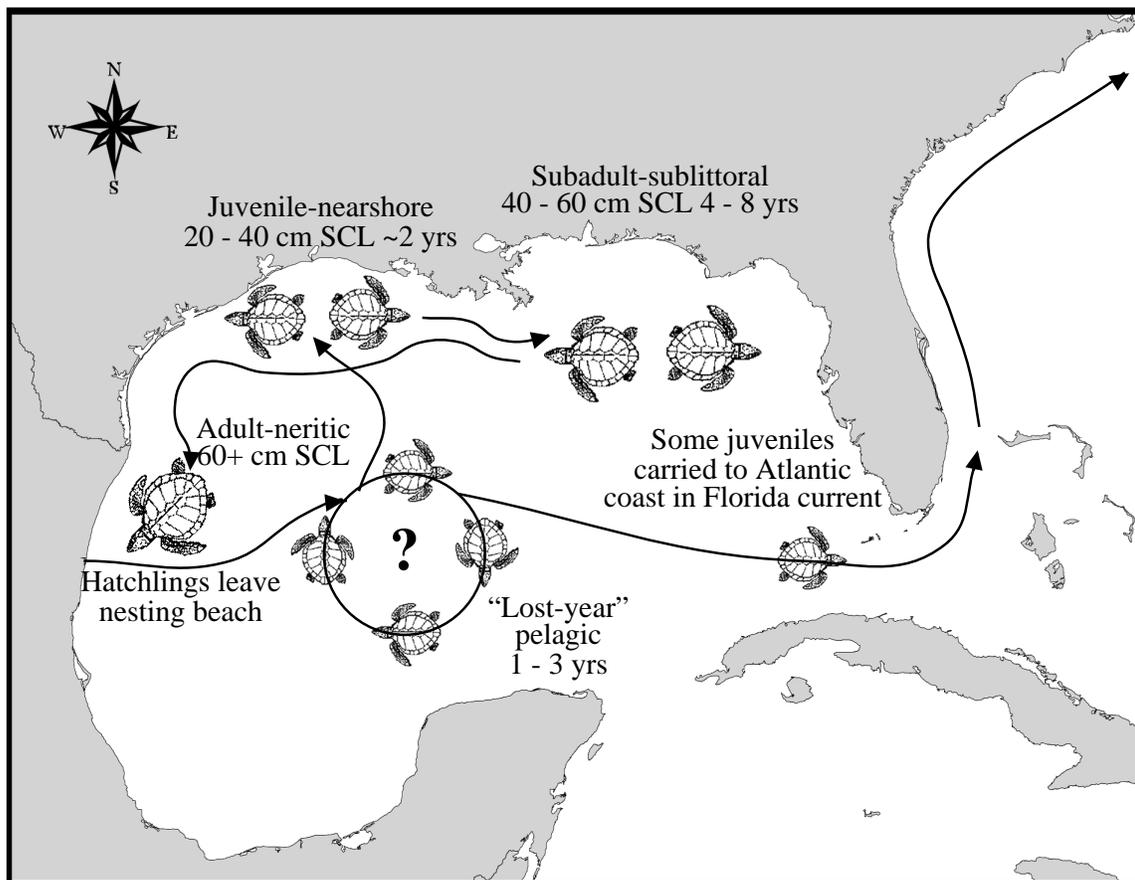


Figure 2. Generalized life history of the Kemp’s ridley sea turtle, *Lepidochelys kempii*.

(3 from Florida and 1 from Chesapeake Bay) suggests these cohorts may indeed be a viable part of the population (Witzell, 1998). Post-hatchling *L. kempii* dispersal is likely related more to movement of surface water masses and wind drift than to swimming.

Carr (1986) speculated that post-hatchling *L. kempii*, as well as those of other species, inhabit the fronts and convergence zones of the Gulf of Mexico and western Atlantic. It has been suggested that young *L. kempii* remain in this pelagic habitat, feeding in convergence zones, until reaching approximately 20 cm SCL (Ogren, 1989). Skeletochronological analysis of circa 20 cm SCL individuals stranded along the Atlantic coast suggests that post-hatchling *L. kempii* remain in this pelagic stage for approximately 2 years (Zug, 1990; Chaloupka and Zug, 1997; Zug *et al.*, 1997).

**Pre-adult Coastal:** Upon reaching approximately 20 cm SCL, *L. kempii* move into the coastal zone (Landry *et al.*, 1993; 1995; 1996) and become benthic feeders, preying primarily on portunid crabs (Werner, 1994). Ogren (1989) defined these coastal-benthic *L. kempii* as subadults ranging from 20 to 60 cm SCL.

Testosterone (T) data suggest these coastal-benthic individuals can be further subdivided into prepubescent and pubescent constituents, or juvenile and subadult, respectively (Coyne and Landry, 2000). Kemp's ridleys from 20 to 40 cm SCL exhibit a low plasma T concentration, indicating little or no reproductive development, and, as such, are considered prepubescent (Wibbels, 1988). Pubescent *L. kempii* exhibit an apparent exponential increase in circulating plasma T corresponding with size after approximately 40 cm SCL, characteristic of a period of reproductive development.

Telemetric tracking of *L. kempii* suggests those less than 40 cm SCL inhabit nearshore (shore to 10 meters) environs while most larger conspecifics remain farther from shore, becoming more sublittoral (mean low tide to about 200 m depth) or neritic (mean low tide to the continental shelf). Tracking data collected by Renaud *et al.* (1995) indicate that *L. kempii* (n=40) less than 17 kg (approx. 26 - 50 cm SCL) exhibit a mean

distance from shore of 4.9 km while those exceeding 24 kg (> 55 cm SCL) averaged 17.4 km from shore. In addition, these workers reported that smaller *L. kempii* exhibited much less movement (< 50 km) than did larger cohorts (up to 2,000 km). These data may indicate a shift in habitat utilization (i.e. distance from shore) or movement patterns as *L. kempii* progress from the juvenile to subadult stage.

**Adult Neritic:** Kemp's ridleys reach sexual maturity at approximately 60 cm SCL (Márquez, 1994). Like juveniles and subadults, adult *L. kempii* are also neritic benthic-feeders. However, depth-at-capture data for adults suggest they inhabit deeper waters than do their pre-adult cohorts (Ogren, 1989). Recent tracking data also indicate that adults range over a much wider area than do younger nearshore conspecifics (Renaud *et al.*, 1995, 1996).

Water temperature is most likely the greatest factor influencing large-scale movement patterns of neritic *L. kempii* (Ogren, 1989). Shallow coastal waters of the northern Gulf of Mexico serve as foraging habitat during warmer months. Cold winter temperatures may cause *L. kempii* to move first offshore, into deeper, warmer waters, then south along the coast (Renaud *et al.*, 1995, 1996).

### **Kemp's Ridley Conservation**

The only recorded *L. kempii* "arribada" prior to 1966 was filmed in 1947 by a Mexican engineer, Andres Herrera. This film remained unknown to the scientific community until 1961 when it was shown during the American Society of Ichthyologists and Herpetologists meeting at the University of Texas (Rostal, 1991). During the 1947 "arribada", 40,000 *L. kempii* were estimated to have nested along a secluded stretch of beach near Rancho Nuevo, Mexico (Carr, 1963; Hildebrand, 1963). Reported number of nesting females suffered a massive decline through 1985, but have since exhibited a slow but steady recovery. On 23 May 1968, the number of turtles nesting in a single

arribada had declined to an estimated 5,000 females (Pritchard, 1969), continuing to a low of approximately 230 to 350 adult females (702 nests) in 1985 (Fig. 3). The nesting population has exhibited a modest turn-around since 1985, increasing approximately 11% per year (TEWG, 1998), but remains far below the number of nests estimated for one day in 1967!

### Nesting Beach

The Mexican government declared the Rancho Nuevo nesting beach a national refuge in 1965 (Márquez, 1994). Before that time, over 90% of the nests were

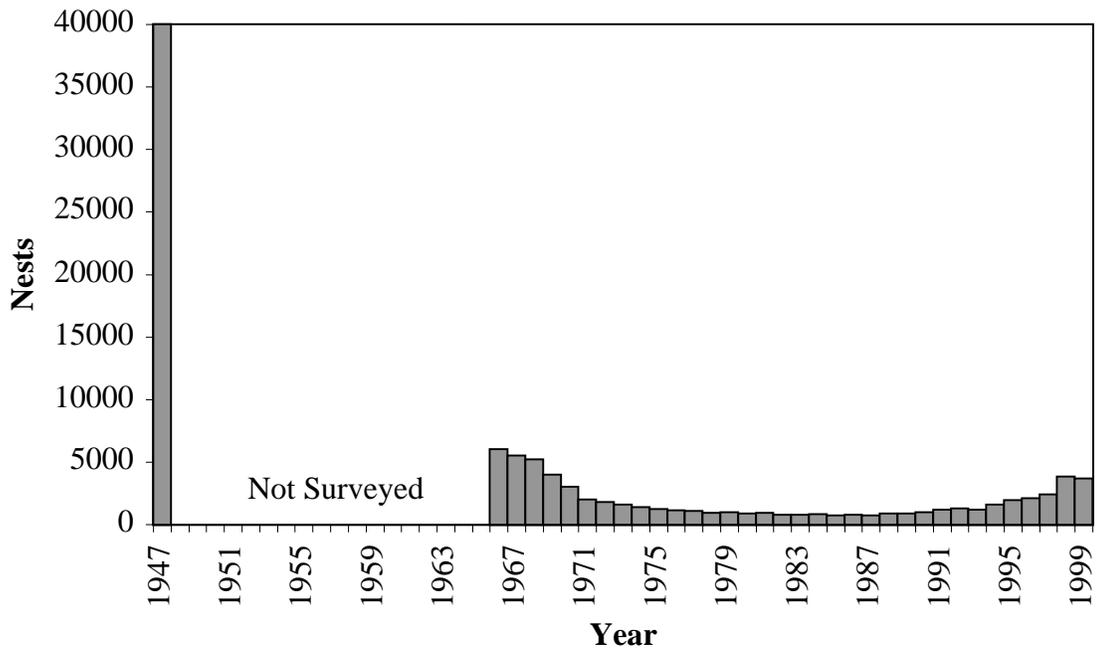


Figure 3. Number of Kemp's ridley nests counted at Rancho Nuevo, Mexico during 1947-1999. The 1947 estimate represents a single day of nesting. Nests recorded since 1978 are from the joint U.S./Mexico Kemp's ridley project while prior numbers were estimated from scattered nesting data (Hildebrand, 1963; Pritchard, 1969; Pritchard and Márquez, 1973).

depredated by man and animals, resulting in near zero recruitment to the adult population for almost three decades. In 1966, measures were taken to stop egg poaching and a research and conservation program was established. Protection of nests, eggs and hatchlings resulted in the release of approximately 30,000 hatchlings per year until 1978.

The Kemp's ridley was initially listed as endangered in the U.S. in 1970 under the Endangered Species Conservation Act of 1969 (Public Law 91-135). The species received additional protection under the U.S. Endangered Species Act of 1973, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES), and various laws, regulations, decrees and acts promulgated by Mexico (Márquez, 1994). A bi-national, multi-agency Kemp's Ridley Working Group composed of the Instituto Nacional de la Pesca, U.S. Fish and Wildlife Service, U.S. National Marine Fisheries Service (NMFS), U.S. National Park Service, Texas Parks and Wildlife Department (TPWD), and university researchers was formed in 1978 to increase protection at the nesting beach and promote development of a second nesting beach at Padre Island National Seashore (PINS), Texas.

### **Headstart**

Approximately 2,000 Kemp's ridley eggs were transported annually from Rancho Nuevo, Mexico to PINS between 1978 and 1988 in an experimental attempt to establish a second nesting beach. These eggs were incubated in Padre Island sand and the resultant hatchlings "imprinted" in water off Padre Island (Owens *et al.*, 1982; Shaver, 1989). Hatchlings were then transported to the NMFS Laboratory in Galveston, Texas where they were "headstarted" for approximately one year before release (Klima and McVey, 1981).

The "imprinting" experiment was terminated in 1988 after which time *L. kempii* were only headstarted. Approximately 2,000 eggs or hatchlings were taken to

Galveston each year where they were raised for 9 - 12 months and then released. This experiment was terminated in 1992 because of a lack of positive results. Criteria for headstarting success included: 1) producing juvenile Kemp's ridleys capable of joining the natural, wild populations, finding their way to nesting beaches, and producing viable offspring of their own; and 2) demonstrating equivalent or superior biological fitness when compared to that of wild conspecifics (Eckert *et al.*, 1994). Until 1992, no headstarted or imprinted ridley had been recorded nesting at PINS or anywhere else. However, two of six *L. kempii* nesting at PINS during 1996 were confirmed as headstarts belonging to the 1986 and 1987 year classes (Shaver, 1996). Nine *L. kempii* were confirmed to have nested at PINS during 1997, none of which were identified as headstart, and another 13 during 1998, four of which were confirmed headstart turtles (Shaver and Caillouet, 1998). Seven nests were laid by four different *L. kempii* confirmed as headstarted individuals, out of 16 reported nests at PINS during 1999 (Shaver, 1999). In addition, headstarted *L. kempii* captured during this study appear well adjusted in the wild, exhibiting a diet similar to that of their wild cohorts (Werner, 1994).

### **At-Sea Mortality**

Incidental capture of *L. kempii* by the commercial shrimp fishery has been identified as a major obstacle to complete recovery of the species (Shaver, 1995). This conclusion has been developed through correlation of sea turtle strandings with closure of the shrimp fishery in Texas territorial waters (Fig. 4). The purpose of this closure is to delay harvest of small brown shrimp emigrating from Texas bays so as to yield a larger, more marketable size and reduce waste of smaller shrimp. TPWD monitors shrimp stocks and is charged with monitoring the closure in Texas' Territorial Sea (out to 9 nautical miles), usually occurring between 15 May - 15 July. The Gulf of Mexico Fisheries Management Council has simultaneously closed federal waters off Texas to

coincide with Texas closure since 1981. Typically, strandings decrease during closure and rise after shrimping resumes (Fig. 4).

Attempts to regulate the shrimp fishery with respect to sea turtle protection began in 1978 and have ranged from resuscitation requirements for incidentally captured sea turtles to implementing turtle excluder device (TED) technology in the shrimp fishery. First introduced in a voluntary program in 1983 (48 FR 39276), NMFS advocated the benefits of TED use (reduced by-catch, sorting time, and increased fuel efficiency), and introduced regulations requiring their use in all U.S. Gulf of Mexico and South Atlantic waters by 1990 (Public Law 100-416; Public Law 100-478). Currently, shrimpers are required to use hard TEDs, purportedly more effective at releasing sea

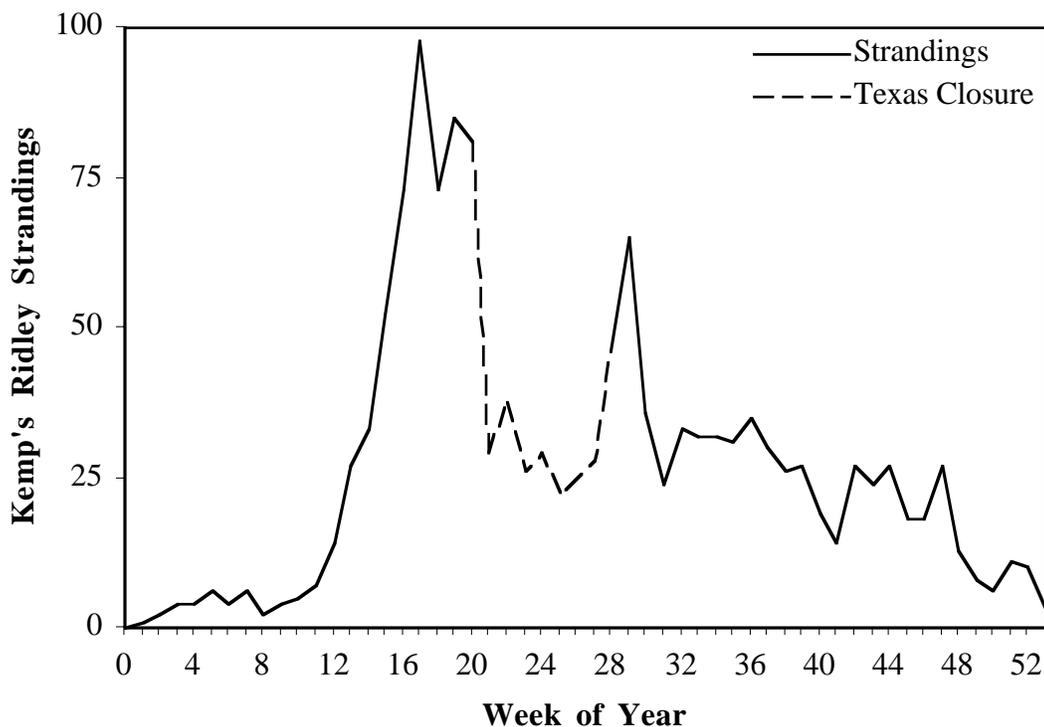


Figure 4. Kemp's ridley strandings reported along the Texas coast during 1986 - 1996. Dotted-line indicates stranding during the Texas closure. Data from NMFS Sea Turtle Stranding and Salvage Network, Miami Laboratory.

turtles entrained in their nets. Hard TEDs also are required in large trawl nets (61 FR 66933).

Despite these rules, Kemp's ridley strandings have not decreased in the Gulf of Mexico. It is unclear whether this is due to a greater number of turtles at risk of being caught or ineffectiveness of protection efforts. Additional potential sources of Kemp's ridley at-sea mortality include other fishing methods (e.g., long lines, gill nets, purse seines), ingestion of marine debris, explosive removal of offshore oil platforms, oil spills, and other pollution sources.

### **Embryonic Sex Determination**

The influence of temperature on sea turtle embryonic sex determination presents an interesting puzzle. Temperature-dependent sex determination (TSD), also known as environmental sex determination (ESD), acts in sea turtles to produce female hatchlings at warm temperatures and males at cool temperatures. The sensitive period for sex determination appears to occur around the middle third of incubation (Yntema and Mrosovsky, 1982). The threshold or pivotal temperature for transition from production of one sex to the other (i.e., a sex ratio of 1:1) is between 28 to 30° C for all species for which data are available.

In addition to temperature, administration of exogenous estrogen can influence sea turtle sex ratios. Application of estrogen to an egg incubating at a male-producing temperature can reverse the effect of temperature and result in a female hatchling (Raynaud and Pieau, 1985; Gutzke and Bull, 1986; Bull *et al.*, 1988; Crews *et al.*, 1989, 1991; Wibbels *et al.*, 1991a, 1991b; Tousignant and Crews, 1994). This provides a potential method for obtaining female offspring without regard to incubation temperature.

TSD and sex ratios have an interesting history with regard to Kemp's ridley

recovery efforts. During 1978-1988 eggs were collected from Rancho Nuevo in plastic bags as they were laid, packed in Styrofoam™ boxes containing sand from the Padre Island National Seashore (PINS) near Corpus Christi, Texas, and transferred to the PINS for incubation and hatching in the same boxes (Shaver *et al.*, 1988; Burchfield and Foley, 1989). Before 1985, the pivotal temperature for incubation of *L. kempii* eggs was not known (Shaver *et al.*, 1988; Fletcher, 1989), and seems not to have been considered despite published reports of TSD in other sea turtle species at the time (Mrosovsky and Yntema, 1980; Yntema and Mrosovsky, 1980, 1982; Miller and Limpus, 1981; Ruiz *et al.*, 1981; Morreale *et al.*, 1982; Mrosovsky, 1982).

Concern arose regarding the Kemp's ridley program adversely effecting sex ratios (Mrosovsky, 1985) and incubation temperatures were intentionally raised in 1985 in an effort to increase the proportion of females emerging from eggs incubated at PINS (Shaver *et al.*, 1988; Fletcher, 1989). Reviewers later concluded that most samples from year-classes through 1984 were male-dominated and those from the 1985-1992 year-classes were female-dominated (Shaver *et al.*, 1988; Wibbels *et al.*, 1989; Caillouet, 1995). Estimates of female percentage from each year-class are admittedly biased (Wibbels *et al.*, 1989), but the shift to a female-bias was probably real (Caillouet, 1995).

A review of incubation temperatures in relation to percent females produced from 1982 -1987 clutches (n = 32) estimated that the pivotal temperature for *L. kempii* was 30.2° C, with temperatures above 30.8° C producing 100% females (Shaver *et al.*, 1988). In addition to the small sample size, the variation within species is not well defined and other factors have been reported to effect embryonic sex determination (Ackerman, 1996). Bull (1985) found that the variance of incubation temperature, as well as mean temperature, influenced sex determination in *Graptymes* sp. Standora and Spotilla (1985) stated that osmotic stress and O<sub>2</sub> and CO<sub>2</sub> levels may influence sex within the critical range where a mixture of males and females are typically produced.

Limpus *et al.* (1985) found that pivotal temperature in *Caretta caretta* may vary from clutch to clutch. In any case, one must assume that pivotal temperatures produced by relatively constant incubator conditions can be extrapolated to the nesting beach, particularly when using nest temperature as an indicator of sex (Wibbels, 1998; Wibbels and Geis, 1999).

A problem with this assumption is that little is known about the long-term variation of beach temperature and its effect on sex ratios (Mrosovsky, 1994). A limited study of beach temperature profiles during 1986 at Rancho Nuevo and PINS suggested, based upon the reported pivotal temperature, that clutches undergoing the middle third of incubation early in the season should produce primarily males, a mixture at mid-season, and primarily females late in the season (Standora and Spotilla, 1985; Shaver *et al.*, 1988). However, considerable temperature variation can be expected within a nesting beach over time. In addition, local variations in nesting beach temperature, as a result of vegetative cover, local weather conditions and embryonic metabolic heat in the nest, should be taken into account.

Some authors have advocated the use of TSD or administration of exogenous estrogen to produce more females as a conservation tool in the recovery of endangered or threatened turtle populations (Tousignant and Crews, 1994; Vogt, 1994). This suggestion is based on the assumption that female turtles are more important than males since one male can inseminate many females. Captive breeding programs which produce and release hatchlings at a ratio of 6-20 females for every male have been suggested as a recovery strategy for rare and declining populations (Vogt, 1994).

Other authors have cautioned against manipulating hatchling sex ratio, suggesting that an understanding of the influence of sex ratio manipulation and natural sex ratio variation in sea turtles is needed (Mrosovsky and Godfrey, 1995; Lovich, 1996). In general, adult sex ratios in turtle populations can vary according to several

underlying factors (Gibbons, 1990; Lovich and Gibbons, 1990) including sex-specific mortality (Gibbons, 1968; Parker, 1984, 1990), immigration and emigration (Parker, 1984; Gibbons *et al.*, 1990; Lovich, 1990), growth (Chaloupka and Limpus, 1997; Limpus and Chaloupka, 1997), and maturation time (Gibbons, 1990; Gibbons and Lovich, 1990; Lovich *et al.*, 1990).

### **Study Objectives**

Human population increases and habitat degradation have driven many historically-abundant species, like the Kemp's ridley, toward extinction. Correspondingly, attempts to preserve rare or endangered taxa through proactive management have been compromised by decisions based more on ease of implementation or accessibility to particular life stages than clear expectations of population responses to management (Crouse *et al.*, 1987). Nearly all sea turtle conservation efforts have focused on two, easily accessible, life history stages: eggs and adult females on the nesting beach.

Models which generate meaningful output for management decisions are based on input realistically describing the life history of the population. Unfortunately, many parameters essential to realistic sea turtle models (i.e., survivorship, age-at-maturity and longevity) are difficult to quantify (Bustard, 1979). Other components critical to developing useful population models include many characteristics related to reproduction, such as adult female remigration rate, number of eggs per nest, seasonal nesting frequency, and sex ratio.

Information on the sex ratio of wild animal populations is useful for population modeling and provides a baseline against which to assess conservation strategy (Mrosovsky, 1994). As an example, TSD in sea turtles provides a potential recovery tool for these threatened and endangered species (Mrosovsky, 1980; Morreale *et al.*, 1982;

McCoy *et al.*, 1983; and Shaver *et al.*, 1988). Resource managers may choose to masculinize, feminize or balance a population's gender by manipulating incubation temperatures (Shaver *et al.*, 1988). While the ability to control sex of hatchlings is advantageous, choosing the appropriate gender option without knowledge of the natural sex ratio and how subsequent alterations might impact population balance is difficult.

Unfortunately, sea turtle sex can be difficult to determine. No evident secondary sexual characteristics develop until maturity, at which time the male's plastron becomes soft and the tail extends beyond the carapace (Wibbels *et al.*, 1991a; Owens, 1997). Consequently, the only reliable method of sexing immature animals is through direct observation of the gonads (Van Der Heiden *et al.*, 1985; Mrosovsky and Benabib, 1990) which requires sacrificing the turtle, use of surgery (i.e., laparoscopy) or necropsy of stranded carcasses (Shaver, 1991; Stabenau *et al.*, 1996). Sacrifice is not a viable alternative for an endangered species and, while laparoscopy has proven reliable and mostly harmless (Limpus *et al.*, 1994a), sample size requirements and field restrictions under endangered species regulatory framework make dependence on this procedure impractical. Finally, advanced decomposition of most stranded sea turtles often renders sex determination difficult or unreliable (Heinly, 1990).

This study utilized radioimmunoassay (RIA) determination of blood plasma T concentration in conjunction with limited laparoscopy to sex Kemp's ridley sea turtles. These techniques have been used concurrently in other studies to successfully sex both loggerhead (*Caretta caretta*, Wibbels *et al.*, 1987) and green (*Chelonia mydas*, Bolten *et al.*, 1992) sea turtles. Sexing criteria were established by examining plasma T concentration of turtles whose sex had been verified via laparoscopy. These criteria were then applied when sexing other turtles from which only a blood sample had been obtained and plasma T measured. These results and information available in the scientific literature were used to develop a preliminary population dynamics model for *L.*

*kempii*. Model development highlighted areas of insufficient knowledge or lack of understanding concerning *L. kempii* population dynamics (Grant, 1986). The final model was used primarily to assess the impact of various population sex ratio values upon Kemp's ridley sea turtle demography.

### **Objectives**

A quantitative study of blood plasma testosterone in Kemp's ridley sea turtles (*Lepidochelys kempii*) from the northwest Gulf of Mexico was conducted to:

1. Establish testosterone radioimmunoassay sexing criteria for *L. kempii*;
2. Estimate a sex ratio for the wild *L. kempii* population;
3. Develop a preliminary population model for *L. kempii*; and
4. Assess impact of different sex ratio values on *L. kempii* population dynamics.

## CHAPTER II

### SEXING KEMP'S RIDLEY

*Lepidochelys kempii*, like other sea turtle species, is heterosexual, exhibiting sexual dimorphism in adults (Márquez, 1994). These differences become most evident as secondary sexual characteristics of subadults in the final phase of maturation and in adults. Mature males have a large tail which extends well beyond the carapace, and a strongly curved flipper claw used in holding the female during copulation (Owens, 1997). Mature females do not exhibit secondary sexual characteristics, but typically bear scratches and scars on the anterior edge of the carapace caused by the male during copulation.

Some pubescent or sexually developing turtles might be identified as males by their softening plastron; however, such identification requires the most experienced observer (Wibbels *et al.*, 1991a). The function of the softened male plastron has not been investigated, but may facilitate mounting during copulation (Owens, 1997). A method, both simple and reliable, for sexing immature sea turtles has yet to be identified. Techniques commonly employed to assess sex in immature sea turtles include: histological analysis of gonads, gross examination of stranded carcasses, H-Y antigen assay, laparoscopy, and radioimmunoassay (RIA).

Histological examination, used most frequently to sex hatchling sea turtles, involves removing the gonads and subsequent microscopic examination (Mrosovsky *et al.*, 1984). This technique requires that subjects be euthanized, an undesirable consequence when studying threatened or endangered sea turtles. In addition, this methodology can be quite laborious (Mrosovsky and Benabib, 1990).

Necropsy and observation of gonads also have been used to determine sex of stranded Kemp's ridleys (Danton and Prescott, 1988; Shaver, 1991; Morreale *et al.*, 1992; Stabenau *et al.*, 1996). While it is desirable to obtain all available data from every

stranded sea turtle, sex determination can be difficult or unreliable, particularly in severely decomposed carcasses (Heinly, 1990; Owens, unpub. report). For example, gonadal decomposition resulted in sex being determined in only 50% of the stranded sea turtles examined by Stabenau *et al.* (1996) from the upper Texas coast. In addition, stranded turtles may not provide a representative subsample of the entire population.

Detection of the H-Y antigen also has been used to sex green and loggerhead sea turtles (Wellins, 1987; Foley, 1994). This methodology uses fluorescence microscopy, X-ray film or a cytotoxicity assay to detect the male-specific cell surface histocompatibility antigen. Results have been consistent with the pattern of H-Y positive males found in most other vertebrates, but have not been confirmed using other methods.

One reliable method for determining sex and reproductive status in sea turtles involves laparoscopy to examine the gonads and associated reproductive ducts (Wibbels, 1988; Limpus *et al.*, 1994a). Laparoscopy has been used to assess sex, maturity, and breeding status in *C. mydas* (Limpus and Reed, 1985), *C. caretta* (Limpus, 1985; Wibbels *et al.*, 1987), and *Eretmochelys imbricata* (Limpus, 1992; van Dam, pers. comm.). However, this procedure requires specialized equipment and training and is difficult to perform in the field (Wibbels *et al.*, 1987, 1991b). Laparoscopy also has been used to validate sex determined via RIA (Wibbels, 1988).

First described for use with sea turtles by Owens *et al.* (1978), RIA has been utilized to assess sex ratios and study reproductive cycles in green (Owens *et al.*, 1978; Bolten *et al.*, 1992) and loggerhead sea turtles (Wibbels *et al.*, 1987, 1990). Unfortunately, very little RIA work has been conducted on *L. kempii*. Three studies involving RIA to sex *L. kempii* have been limited to headstarted (Wibbels *et al.*, 1989), captive (Morris, 1982) and adult turtles (Rostal, 1991), the last focusing on the reproductive cycle. No large-scale study has characterized sex ratio across size classes

in wild *L. kempii*. This chapter investigates the use of testosterone RIA and limited laparoscopy to sex and determine a population sex ratio for wild Kemp's ridleys. Results were evaluated against a hypothetical sex ratio of 1:1.

## Materials and Methods

### Study Areas

Sea turtle capture activities were conducted in jetty and beachfront habitats immediately adjacent to Calcasieu Pass, Louisiana, and Sabine Pass, Bolivar Roads (Galveston), and inshore habitat of Matagorda Bay, Texas from September 1992 through September 1997 (Fig. 5). Sea turtles were captured utilizing one or more stationary entanglement nets set adjacent to one another.

**Matagorda Bay:** Matagorda Bay is a shallow bay (average depth - 2 m) bisected by the

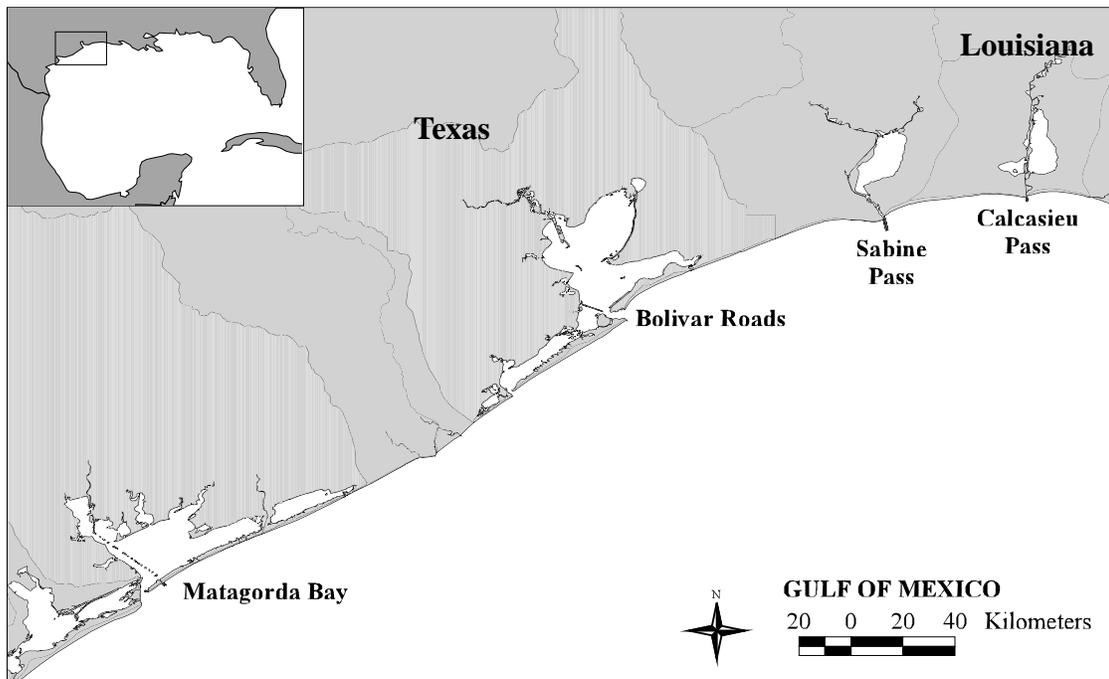


Figure 5. Matagorda Bay, Bolivar Roads, and Sabine and Calcasieu Pass study areas along the northwestern coast of the Gulf of Mexico.

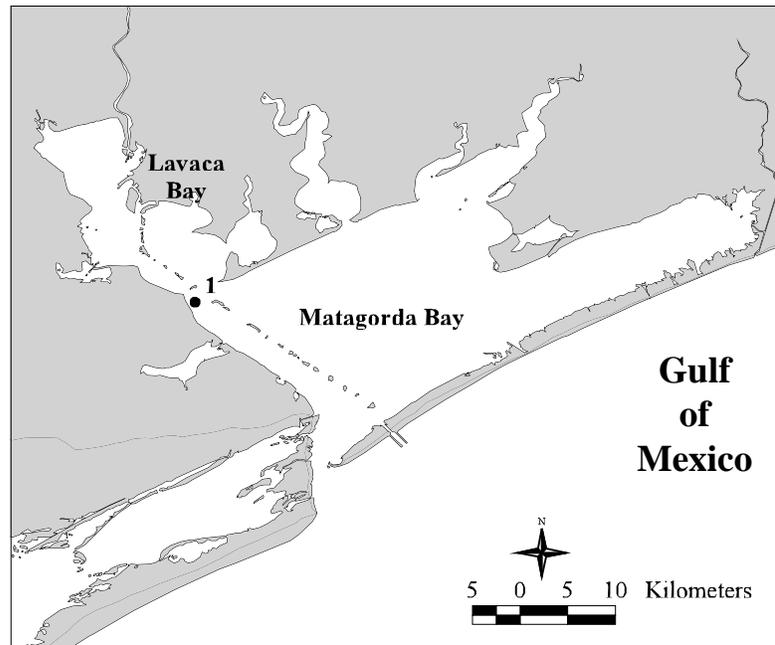


Figure 6. Sea turtle entanglement netting station (•) in the Matagorda Bay study area.

Alcoa Ship Channel just inshore of Matagorda Island and Matagorda Peninsula (Fig. 6). Habitat features include a hard sand/shell hash bottom with scattered seagrasses and oyster beds. One netting station was located near the confluence of Lavaca and Matagorda Bays just off Magnolia Beach, Texas. Entanglement nets were placed perpendicular to the beach and extended to a maximum depth of 2 m. Tidal currents were undetectable to strong. This location was chosen because increasing water depths in other locations prohibited effective netting. The Matagorda Bay station was monitored monthly from May through October 1996.

**Bolivar Roads:** Bolivar Roads is the southernmost portion of the Houston Ship Channel as it enters the Gulf of Mexico between Galveston and Bolivar, Texas (Fig. 7). This deep-water area (maximum depth - 15 m) is protected by north and south jetties which extend gulfward from Bolivar Peninsula on the north and Galveston Island on the south. The jetties are granite mound structures 6.4 km long and approximately 2.1 km apart. Gradual sloping beaches with a hard sand bottom and patches of silt and clay occur

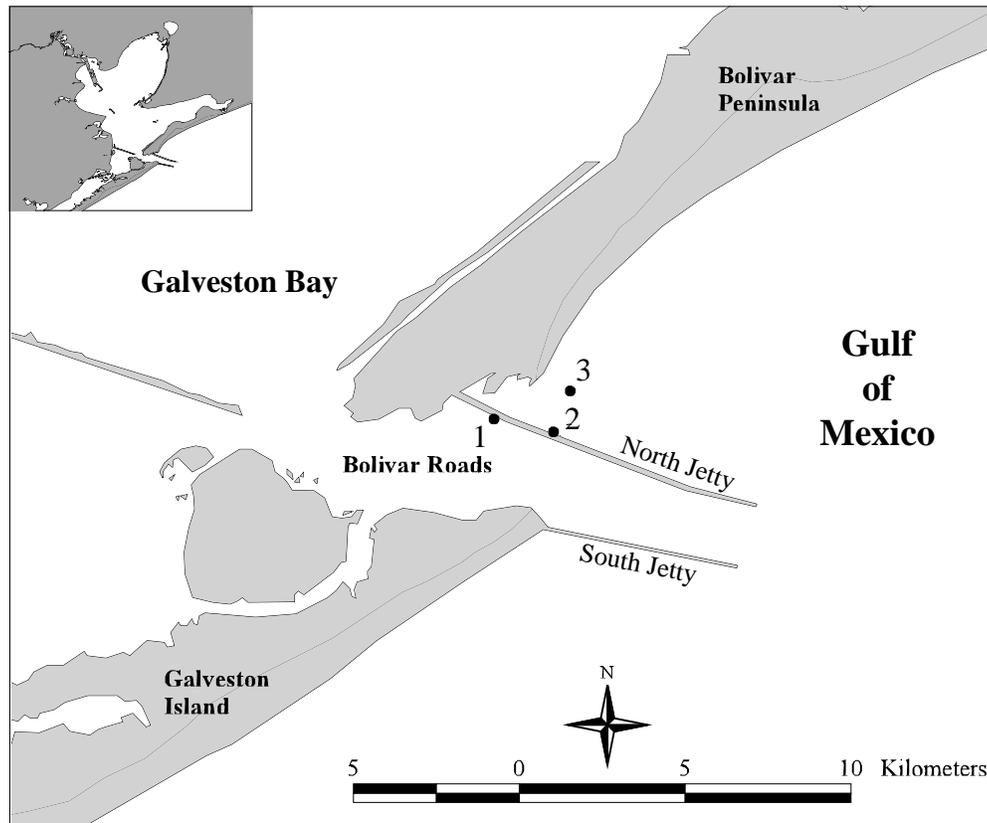


Figure 7. Sea turtle entanglement netting stations (•) in Bolivar Roads study area.

outside each jetty. Habitat features occurring between the north and south jetties include: 1) a mud/clay/sand bottom shelf extending from the jetties to a depth of 5 - 12 m; and 2) the tidally-scoured, dredged channel proper in the Galveston Bay Entrance Channel.

Three netting stations were located within the Bolivar Roads study area (Fig. 7). The immense size of the study area necessitated that each station location be determined, in part, by its accessibility during unfavorable weather conditions. Station 1 was located inside the north jetty about 1.6 km from the Bolivar Peninsula beach in 2 to 3 m of water. This location was chosen because increasing water depths and swift currents farther from shore (and closer to the channel proper) prohibited effective netting at other sites within the jetties. Station 2 incorporated shallow (1 to 3 m) Gulf netting

sites, extending approximately 150 m to either side of a boat cut in the north jetty. Strong surf, swift currents and excessive depths prevented deployment of a monitoring station near the south jetty. Station 3 was a beachfront site approximately 200 m gulfward of the Bolivar Peninsula and 1000 m outside the north jetty in shallow water (1 to 2 m). Bolivar Roads stations were monitored randomly from May 1993 through November 1994.

**Sabine Pass:** Sabine Pass extends from Sabine Lake gulfward 9.66 km to form the southernmost border between Texas and Louisiana (Fig. 8). Granite-mound jetties, 5.6 km long and 503 m apart, border the channelized pass on its east and west sides. Habitat features occurring between the east and west jetties include: 1) a soft mud/clay/sand bottom extending from the jetties to a depth of 2 to 5 m; and 2) the tidally-scoured,

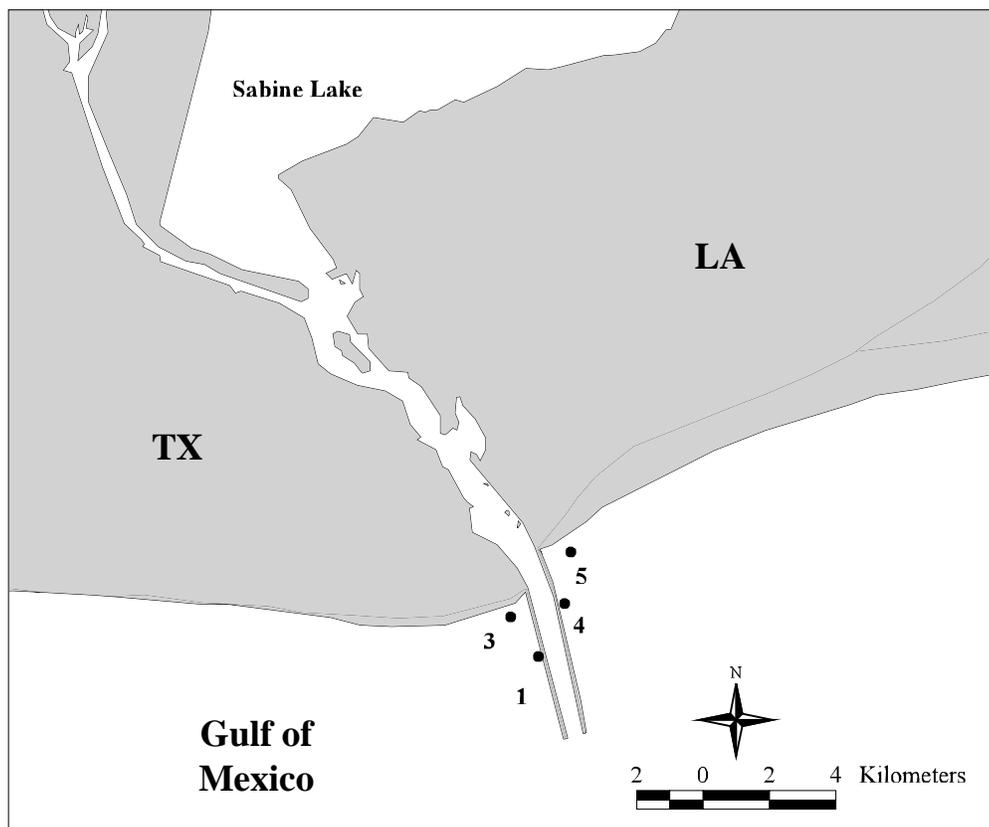


Figure 8. Sea turtle entanglement netting stations (•) in the Sabine Pass study area.

dredged channel proper with a minimum depth of 12 m. Gentle sloping beaches with a hard sand bottom and scattered patches of soft mud and clay occur immediately outside the jetties to a depth of 8 m.

Four entanglement netting sites were established in the Sabine Pass study area (Fig. 8). Two stations (1 and 4) were located immediately adjacent to the outside of the west and east jetties and approximately 1500 and 1200 m from shore, respectively. Water depth at both jetty stations ranged between 1.5 and 3.0 m and tidal currents were undetectable to strong. Stations 1 and 4 were approximately 500 and 300 m north of small boat cuts in the west and east jetties, respectively. Two beachfront stations (3 and 5) were within 1.0 km of each jetty and between 300 and 800 m from shore. Depths at these beachfront stations ranged from 0.6 to 2.0 m and currents rarely exhibited greater than slight tidal flow. Sabine Pass stations were monitored from 1992 through 1997, primarily restricted to April - September.

**Calcasieu Pass:** Located immediately south of Cameron, Louisiana, Calcasieu Pass is the only deep-draft channel between Sabine Pass and the Mississippi River (Fig. 9). The pass is approximately 9.6 km long and has a maximum depth of 13 m. Granite rubble jetties, 1.8 km long and 315 m apart, border the east and west sides of Calcasieu Pass at its entrance to the Gulf. Gentle-sloping beaches with a hard-sand bottom and scattered patches of soft mud and clay occur immediately outside each jetty. Within the jetties, the bottom is composed of soft mud, clay and sand while water depths vary from 2 m at the granite rubble to 13 m in the channel proper.

Four entanglement netting stations (2 jetty and 2 beachfront) were located within the Calcasieu Pass study area (Fig. 9). Two stations (1 and 4) were located immediately outside of the west and east jetties, respectively, and approximately 100 m north of small boat cuts in each jetty. Both jetty stations were approximately 800 m from the beachfront, and each exhibited a maximum depth of 2.5 m and slight to strong

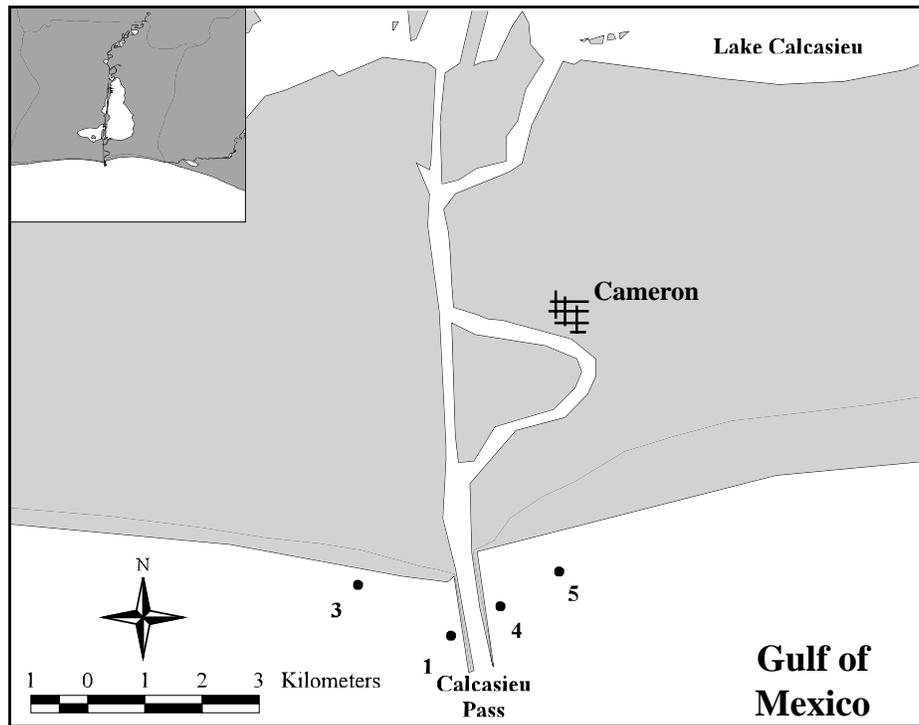


Figure 9. Sea turtle entanglement stations (•) in the Calcasieu Pass, Louisiana study area.

currents. Two beachfront stations (3 and 5) were located 50 to 200 m off the beach and within 800 m of each jetty. These stations exhibited undetectable to occasionally moderate currents and depths of 1 to 2 m. Calcasieu Pass stations were monitored from 1993 through 1995, primarily restricted to April - September.

### Sea Turtle Capture

Sea turtle capture was accomplished with 91.5 m long entanglement nets of different depth and mesh size specifications (3.7 m deep with 12.7 cm bar mesh or 4.9 m deep with 25.4 cm bar mesh, of #9 twisted nylon). Water depth and current dictated which net was used at each monitoring station. All stations were sampled during the day with one or more stationary entanglement nets set adjacent to one another for 6 to 12

hours. Duration of net sets was largely dependent upon weather and sea conditions. Nets were checked every 30 - 45 minutes for the presence of turtles in an effort to minimize capture stress and mortality to these organisms and other by-catch. In addition, any turtles observed in the net between regularly-scheduled checks were removed immediately.

Following capture, all turtles were transported to land-based holding facilities where they were maintained in fiberglass tanks for a minimum of 24 hours before being tagged and released. Captured turtles were monitored periodically to assess health and well being. All turtles were measured, weighed, photographed, and tagged during the holding period.

Straight line carapace length (SCL) and width (to nearest 0.1 cm) were measured with a forester's calipers while corresponding over-the-curve measurements were taken with a nylon measuring tape. Weight was determined (to nearest 0.01 kg) on a 50 kg Tricoastal Industries, Inc. electronic hanging scale. Turtles were tagged with one inconel flipper tag on the trailing edge of each fore flipper and one passive integrated transponder (PIT) tag inserted subcutaneously into the dorsal surface of the right fore flipper. Turtles were scanned for the presence of a previously applied PIT tag before a new tag was inserted. Headstarted *L. kempii* were identified and year class determined by flipper tag, PIT tag and/or living tag. All tagged turtles were released at the site of capture. Tagging data were submitted to NMFS (Miami) on two data forms entitled "NMFS/SEFC Marine Turtle Tagging Data (Rehabilitated, Netted or other Release)" and "NMFS/SEFC Marine Turtle Tagged/Recapture Data".

### **Blood Sampling**

An attempt was made to sample blood from each turtle immediately upon capture (Fig. 10). Initial blood samples were obtained between 7 and 19 minutes ( $\bar{x}$  =



Figure 10. Blood was drawn from the dorsal cervical sinus using a vacuutainer and 22 gauge needle.

12.3  $\pm$  0.2 SE) after turtles were first observed in the net. Bloods samples were taken from the dorsal cervical sinus of each turtle using a 22 gauge needle and vacuutainer (Owens and Ruiz, 1980). These samples were placed on ice and subsequently centrifuged to separate blood cells and plasma for storage in cryogenic vials. Separated samples were again iced, later frozen, and subsequently transported to Texas A&M University for analysis. Plasma testosterone (T) concentration was determined by radioimmunoassay (RIA) in the laboratory of Dr. David Owens as described by Valverde (1996). Assay sensitivity was 0.5 pg/assay tube (Endocrine Sciences, 1972). Intra- and inter-assay variability was 5.64 and 9.77%, respectively.

### **Laparoscopy**

*In situ* observation of gonads was used to verify sex of individual turtles and



Figure 11. Kemp's ridley sea turtle restrained in dorsal recumbency for laparoscopic surgery. Laparoscope was inserted into the abdominal cavity through an incision in the area of the inguinal triangle.

corroborate testosterone RIA results (Wood *et al.*, 1983). Approximately 20% of all captured turtles (74 of 361) were subject to laparoscopic examination of the gonads. One individual exhibiting intersex characteristics was excluded from analysis. To alleviate health concerns, each animal's length-weight relationship, behavior, and general overall condition were noted before laparoscopy commenced.

Each turtle was restrained in dorsal recumbency and the incision area, cranial to the right hind limb, surgically scrubbed and infused with 2% Lidocaine (Fig. 11). A 1-cm stab incision was made with a scalpel through which a 7 mm arthroscopic trocar was inserted into the abdominal cavity. A space to view the gonads was created by partially

inflating the cavity with air from a foot operated pump delivered via a plastic airline connected to the trocar/cannula valve. A laparoscope was then inserted to examine the gonads. Upon completion of this examination, the incision was closed with an absorbable suture and the turtle released no sooner than 24 hours post-surgery. All laparoscopies were performed by Dr. David Owens of Texas A&M University or by the author under Dr. Owens' direct supervision.

### **Statistical Analysis**

Sample means are followed by plus or minus ( $\pm$ ) one standard error. Testosterone data were transformed (natural log) to obtain a normal distribution. Analysis of variance was used for comparison of normalized plasma T values between verified male and female Kemp's ridleys. Sex ratio and distribution of sexes across capture locations, months, size classes, and years were tested against an expected outcome of 1F:1M using  $X^2$ . The JMP statistical package, version 3.2 (SAS Institute, Inc.), was used for all statistical analyses at a significance level ( $\alpha$ ) of 0.05.

## **Results**

### **Sea Turtle Capture**

Entanglement-netting effort at the four study areas resulted in a total of 370 Kemp's ridley captures, including 9 recaptures and 26 headstarted individuals. A total of 618 km-net hours was expended across the study areas. One kilometer-net hour represents each hour that a kilometer of net was set. The majority of effort (69.6%) and captures (307 or 83.0%) occurred at Sabine Pass (Fig. 12). Capture statistics for the remaining sites were: Calcasieu Pass - 16.8% effort and 51 turtles (13.8%); Matagorda Bay - 7.9% effort and 7 turtles (1.9%); and Bolivar Roads - 5.7% effort and 5 turtles (1.4%).

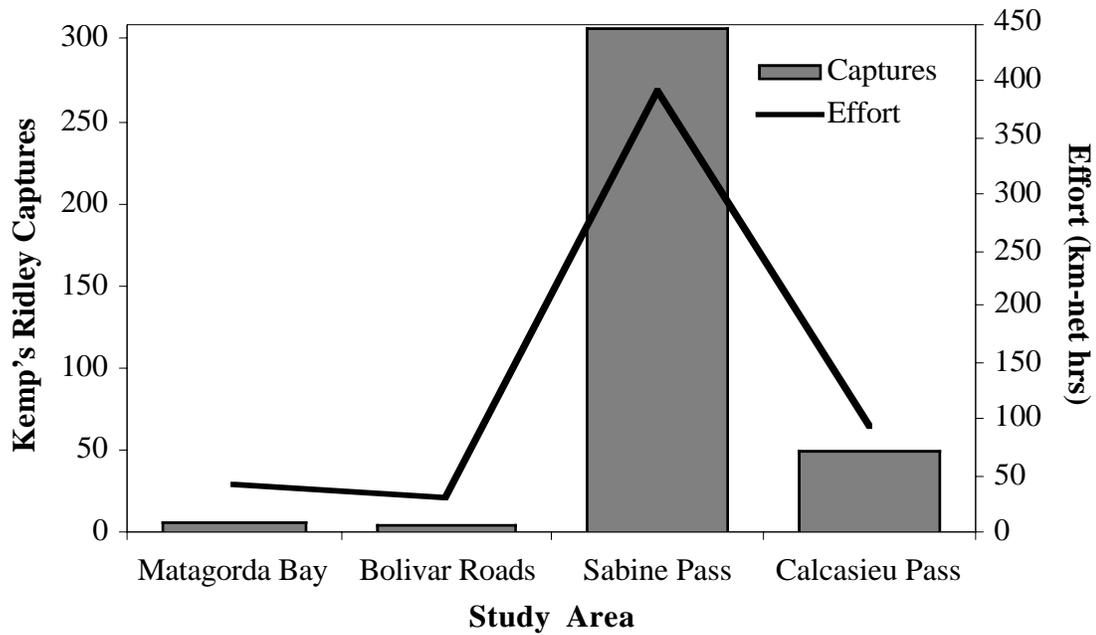


Figure 12. Total number of Kemp's ridley captures versus netting effort (km-net hours) expended at the Matagorda Bay, Bolivar Roads (Galveston), Sabine Pass and Calcasieu Pass study areas.

### Sex Validation

Sex was verified for 79 individual *L. kempii*, including 4 headstarted turtles (3 females, 1 male). Three of these individuals were recaptured once and another twice. In all, 74 individuals were sexed by laparoscopy, three were necropsied by NMFS-Galveston personnel after stranding between 284 and 345 days after capture in the author's nets, and two others were identified as females from flipper tags attached while nesting in Mexico. This yielded overall positive identification of 47 female and 32 male *L. kempii*, with 44 and 31 wild individuals, respectively ( $X^2 = 1.66$ ,  $p \leq 0.1979$ ; Table 1).

Sixty-three capture blood samples (including five recapture samples) were obtained from 58 individual turtles whose sex was verified and included 32 female (with

Table 1. Sex validation results of Kemp's ridley sea turtles subject to laparoscopy, post-mortem necropsy or observed nesting.

Validation Method	N	Total		Recaptures		Head-started		Individuals Bled	
		Female	Male	Female	Male	Female	Male	Female	Male
Laparoscopy	84	47	32	3	2	3	1	32	26
Necropsy	3	1	2	0	0	0	0	1	1
Nesting	2	2	-	0	-	0	-	1	-

three recaptures) and 26 male (with two recaptures) samples. Individuals ranged in size from 21.8 to 62.1 cm SCL for females ( $\bar{x} = 42.8 \pm 12.8$  cm) and 23.9 to 59.2 cm SCL for males ( $\bar{x} = 37.3 \pm 2.3$  cm). These samples were used to set a sexing criteria for all blood samples collected at capture for Kemp's ridleys (Appendix A). Plasma T concentration for sex-verified turtles exhibited a mean of  $4.8 \pm 0.5$  pg/ml for females and  $186.3 \pm 73.6$  pg/ml for males, with respective gender ranges of 0.17 - 12.0 pg/ml and 18 - 2063 pg/ml (Fig. 13). There was a significant difference in plasma T between male and female turtles ( $F = 136.1$ ,  $p \leq 0.0001$ ;  $DF = 59$ ). These data result in a plasma T sexing criteria for *L. kempii* of  $\leq 12$  pg/ml for females and  $\geq 18$  pg/ml for males, with all individuals whose plasma T concentration fell between these criteria designated as indeterminate.

### Blood Sampling

Blood samples were obtained from 247 of 370 (66.8%) *L. kempii* captures (including seven recaptures) across the four study sites. One laparoscoped individual displayed intersex characteristics and was excluded from further analysis. Plasma T RIA results ranged from undetectable to 2063 pg/ml. The calculated sexing criteria applied to all individual *L. kempii* for which capture blood was obtained resulted in 134 females, 95 males ( $X^2 = 5.84$ ,  $p \leq 0.0157$ ) and 10 indeterminates (samples ranging between 12 and

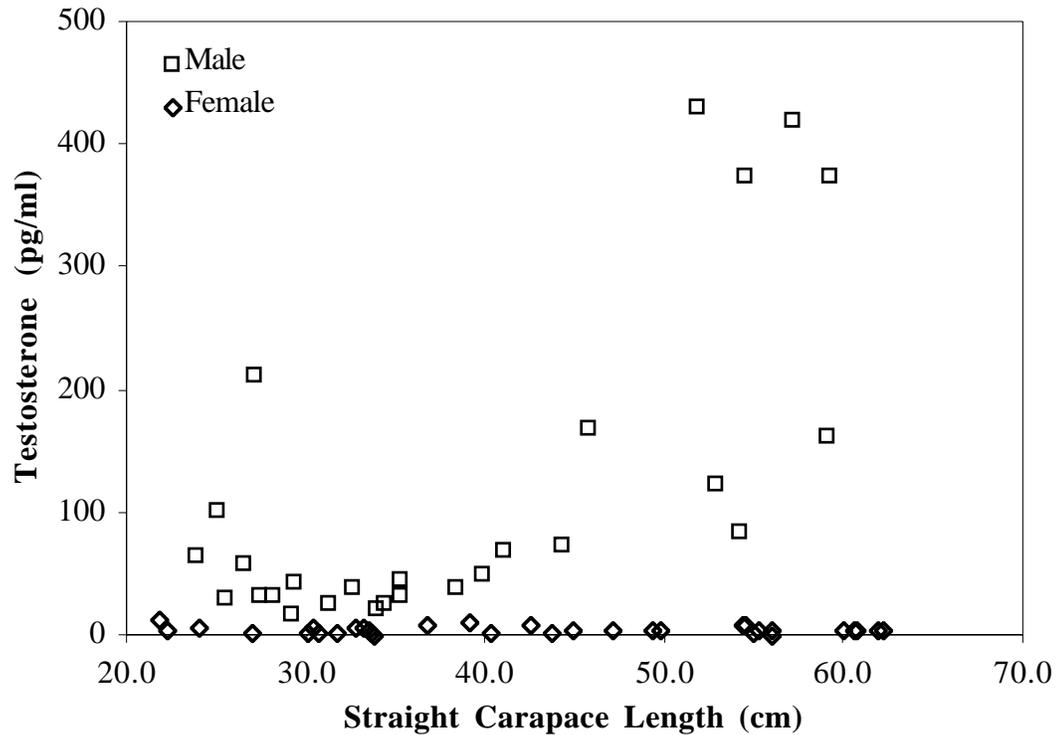


Figure 13. Plasma testosterone concentration (pg/ml) for female (n = 35) and male (n = 28) Kemp's ridley sea turtles whose sex was verified via laparoscopy and necropsy. One 57.9-cm SCL headstart male with plasma T = 2063 pg/ml is not shown.

18 pg/ml). Plasma T RIA results of seven recapture samples, including 5 females and 2 males, were consistent with results from initial capture. Another 16 females and 6 males identified by other means (laparoscopy only, necropsy, or nesting event) increased the experiment lot to 150 females and 101 males. Twenty sexed *L. kempii* were part of the NMFS headstart program including two males and 18 females, resulting in 132 wild females and 99 wild males.

## Sex Ratio

Plasma T RIA data produce a 1.3:1.0 female:male sex ratio for wild *L. kempii* ( $n = 231$ ,  $X^2 = 4.77$ ,  $p \leq 0.0289$ ) and an overall ratio of 1.5F:1.0M when headstart captures are included ( $n = 251$ ,  $X^2 = 9.57$ ,  $p \leq 0.0020$ ). Both ratios are significantly different from 1F:1M. Sex ratio for headstart animals was 9.0F:1.0M ( $n = 20$ ,  $X^2 = 41.99$ ,  $p \leq 0.0001$ ). The proportion of sexed *L. kempii* to the entire capture lot was similar across capture location (Pearson = 0.9990), month (Pearson = 0.9645), year (Pearson = 0.9552) and size class (Pearson = 0.9909) (Fig. 14). Subsequent analyses include only wild *L. kempii* to eliminate confounding effects related to headstart sex ratios.

No significant variation was detected in sex ratio related to capture location for wild *L. kempii* ( $X^2 = 5.64$ ,  $p \leq 0.1304$ ). Sex ratio (Fig. 15) was 1.3F:1.0M for cohorts from Sabine Pass ( $n = 195$ ,  $X^2 = 3.21$ ,  $p \leq 0.0734$ ), 1.8F:1.0M at Calcasieu Pass ( $n = 25$ ,  $X^2 = 1.96$ ,  $p \leq 0.1615$ ), 2.0F:1.0M ( $n = 3$ ,  $X^2 = 0.33$ ,  $p \leq 0.5637$ ) at Bolivar Roads and 1.0F:1.3M at Matagorda Bay ( $n = 7$ ,  $X^2 = 0.14$ ,  $p \leq 0.7055$ ). Sex ratio of *L. kempii* captured within each capture location was not significantly different from 1F:1M. Subsequent analyses exclude *L. kempii* captures from Matagorda Bay, Bolivar Roads and Calcasieu Pass because most captures occurred at Sabine Pass and to avoid potential confounding effects upon sex ratio related to capture location.

Sex ratio of wild *L. kempii* across capture months (primarily May to August) also exhibited no significant variation ( $X^2 = 7.78$ ,  $p \leq 0.3520$ ). Sex ratios for each month with more than 30 captures (May - August) ranged from 1.0F:1.4M ( $n = 34$ ,  $X^2 = 1.06$ ,  $p \leq 0.3035$ ) during August to 1.9F:1.0M ( $n = 40$ ,  $X^2 = 3.60$ ,  $p \leq 0.0578$ ) in July (Fig. 16). Sex ratios during May and June were 1.6F:1.0M ( $n = 62$ ,  $X^2 = 3.16$ ,  $p \leq 0.0754$ ) and 1.2F:1.0M ( $n = 47$ ,  $X^2 = 0.53$ ,  $p \leq 0.4658$ ), respectively. Although May and July exhibited a strong female bias, observed sex ratio was not significantly different from

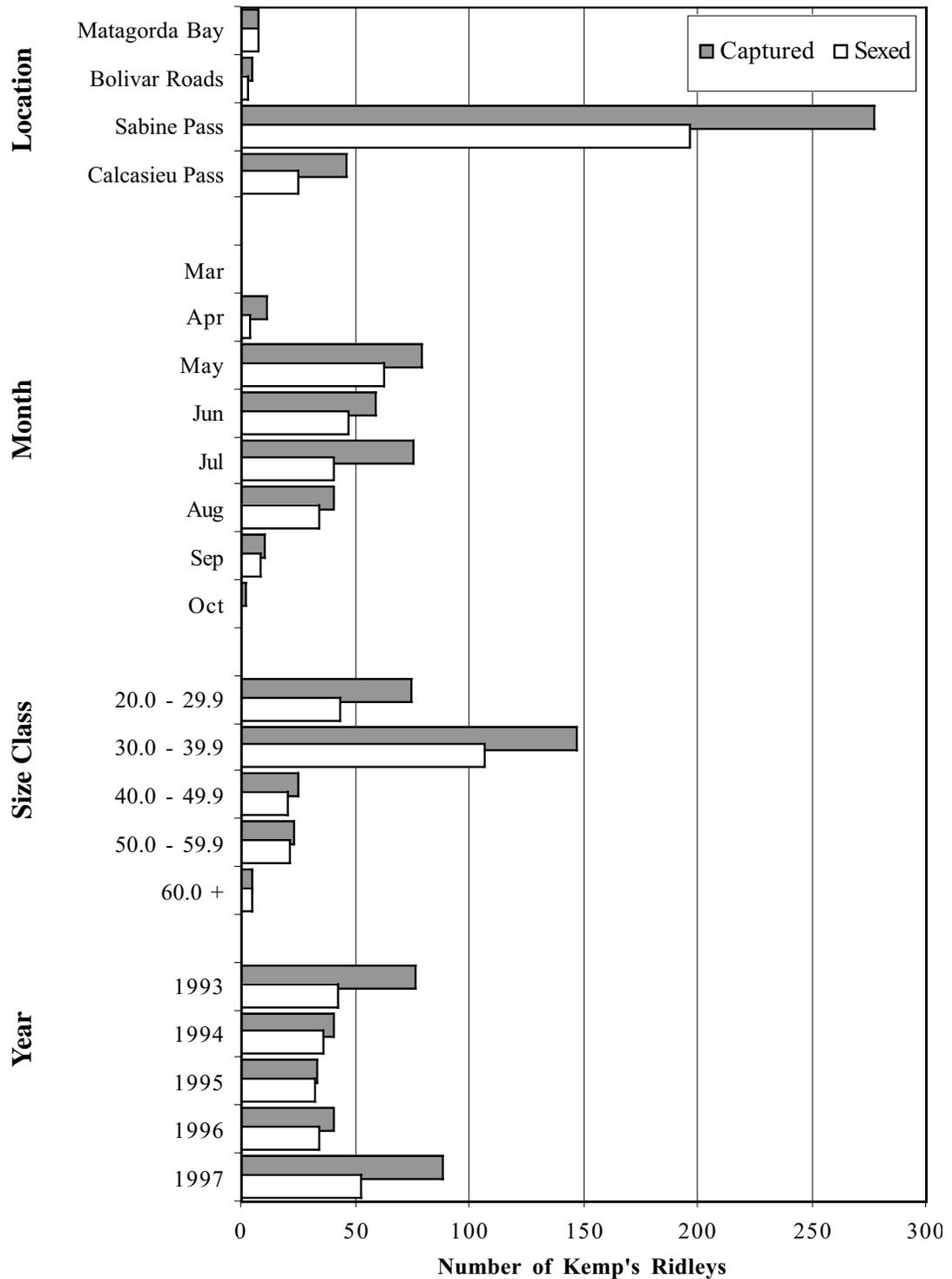


Figure 14. Comparison of all Kemp's ridleys captured during the study to only those that were sexed, across capture location (Pearson = 0.9990), month (Pearson = 0.9645), size class (Pearson = 0.9909) and year (Pearson = 0.9552)

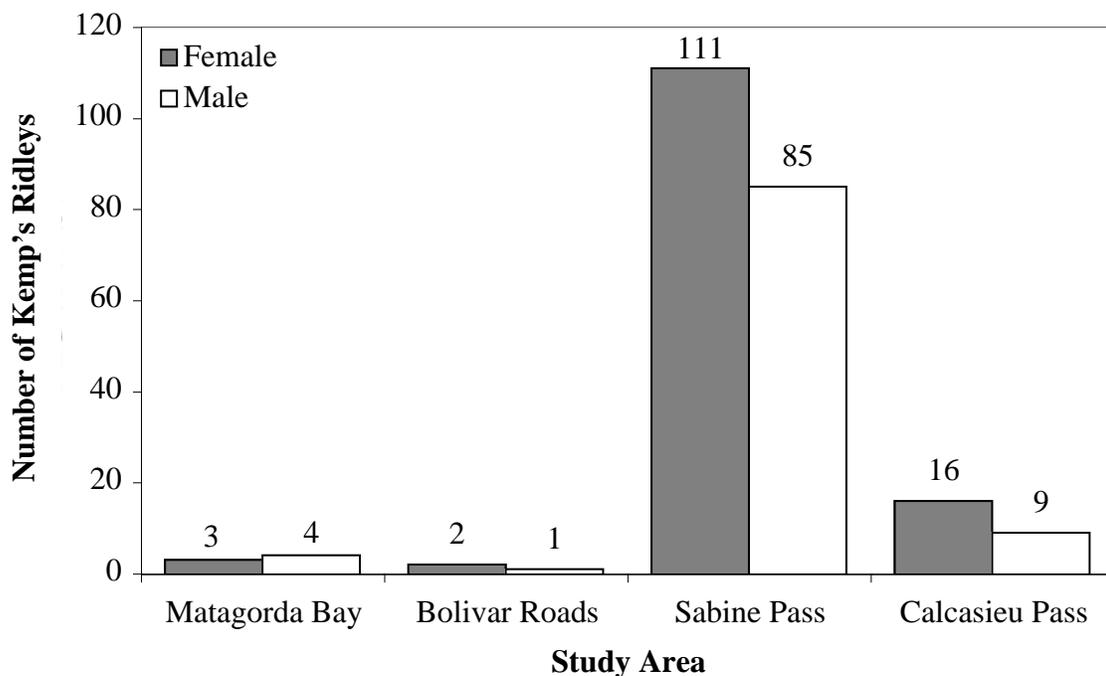


Figure 15. Sex of wild Kemp's ridleys captured from Matagorda Bay, Bolivar Roads, Sabine Pass and Calcasieu Pass study areas during September 1992 - October 1997.

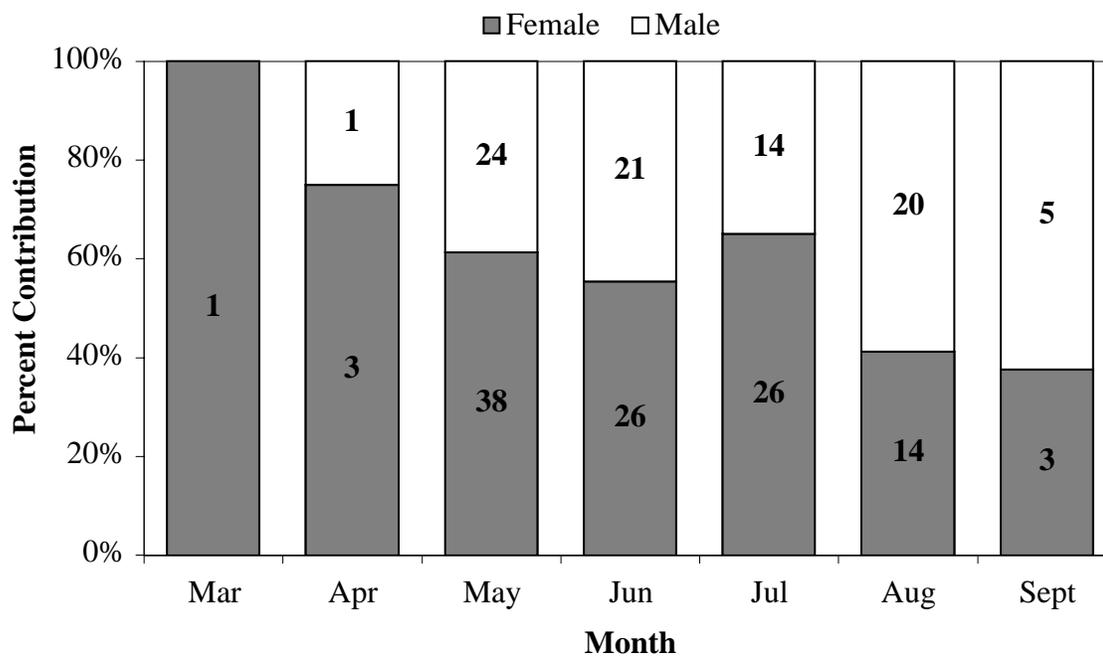


Figure 16. Percent contribution of female and male wild Kemp's ridleys to monthly capture totals during September 1992 - October 1997. Numbers within histogram bars denote sample size.

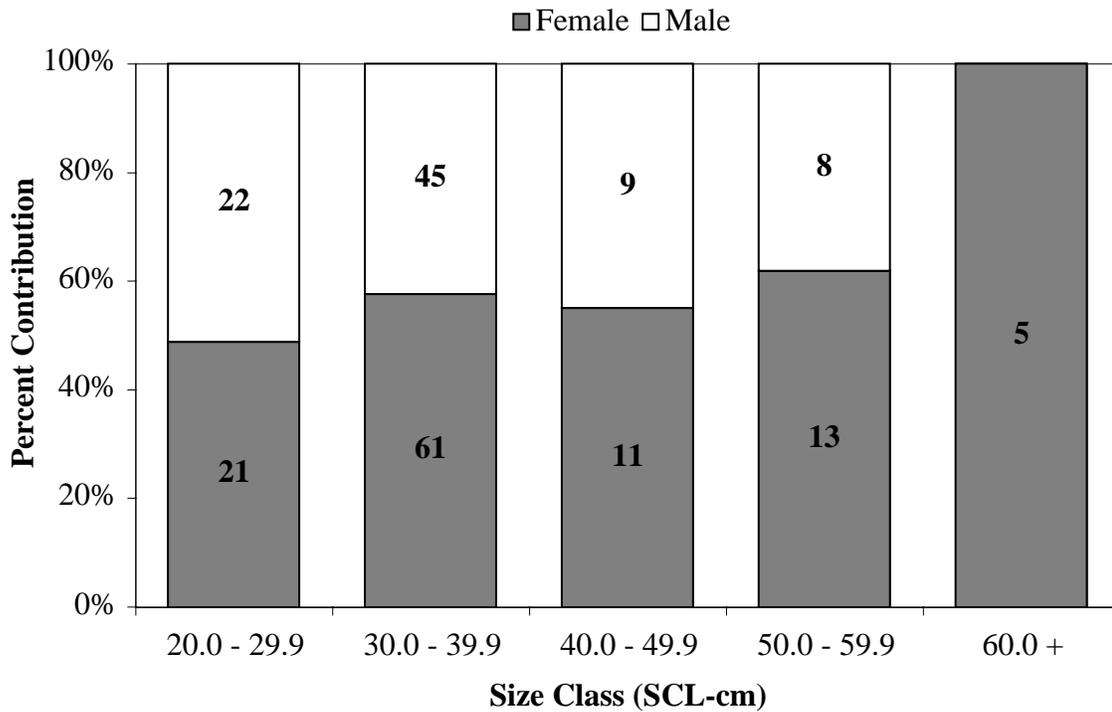


Figure 17. Percent contribution of female and male wild Kemp's ridleys to 10 cm size classes (SCL) captured during September 1992 - October 1997. Numbers within histogram bars denote sample size.

1F:1M during any given month,

No significant variation was observed in sex ratios across arbitrary 10-cm size classes ( $X^2 \leq 8.83$ ,  $p \leq 0.0655$ ). However, larger ridleys did exhibit significant female bias, most obvious in the 60+ cm SCL category ( $X^2 = 5.00$ ,  $p \leq 0.0253$ ) with 5 females and no males (Fig. 17). All turtles  $\geq 50$  cm SCL exhibited a 2.3F:1.0M sex ratio ( $n = 26$ ,  $X^2 = 3.85$ ,  $p \leq 0.0499$ ), while ridleys  $\geq 40$  cm SCL exhibited a sex ratio of 1.7F:1.0M ( $n = 46$ ,  $X^2 = 3.13$ ,  $p \leq 0.0768$ ). Ridleys less than 40 cm SCL yielded a 1.2F:1.0M ratio ( $n = 149$ ,  $X^2 = 1.51$ ,  $p \leq 0.2191$ ). The smallest ridleys (20 - 29.9 cm SCL) exhibited a balanced ratio of 1.0F:1.0M ( $n = 43$ ,  $X^2 = 0.02$ ,  $p \leq 0.8788$ ).

A similar sex distribution across size classes is evident from *L. kempii* stranding records for the U.S. Gulf and Atlantic coasts (Pearson correlation coefficient =

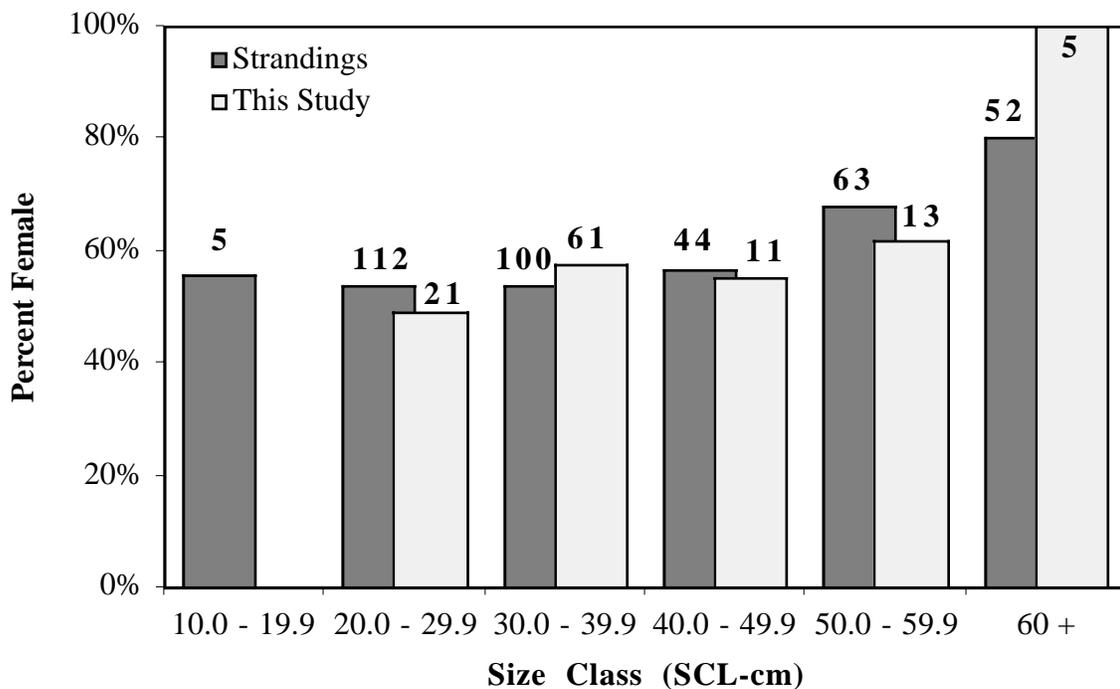


Figure 18. Percent female wild Kemp's ridleys reported stranded and sexed during 1986 through 1996 compared to those captured during this study (Pearson = 0.9232). Numbers within histogram bars denote number of females. Stranding data were obtained from the NMFS Sea Turtle Stranding and Salvage Network, Miami Laboratory.

0.9232). Data were obtained from the NMFS Sea Turtle Stranding and Salvage Network (Fig. 18). Analysis included wild *L. kempii* stranded during 1986 through 1996 along the U.S. Gulf and Atlantic coast for which SCL and sex were recorded. Additional ridleys were included for which sex and curved carapace length (CCL) were recorded by converting CCL to SCL using a regression equation calculated from *L. kempii* captured during this study ( $n = 285$ ;  $r^2 = 0.9920$ ):

$$\text{SCL} = 0.346 + 0.948 \cdot \text{CCL}$$

This equation accurately predicted SCL using CCL from those stranded animals for which both SCL and CCL were recorded ( $n = 1344$ ,  $r^2 = 0.9950$ ).

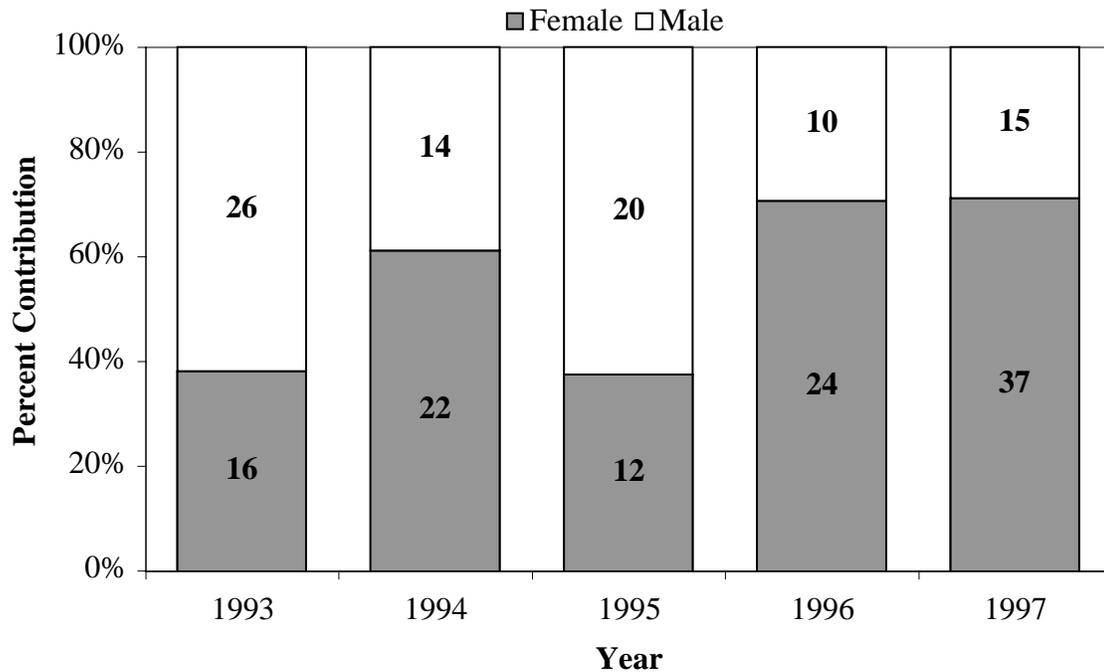


Figure 19. Annual percent contribution of female and male wild Kemp's ridleys during 1993 - 1997. No turtles were sexed during 1992. Numbers within histogram bars denote sample size.

Sex ratio calculated from these stranding records was 1.4F:1.0M ( $n = 584$ ,  $X^2 = 5001.4$ ,  $p \leq 0.0001$ ). Again, gender make-up of smaller size classes was relatively uniform, with ridleys  $<40$  cm SCL exhibiting a sex ratio of 1.2F:1.0M ( $n = 403$ ,  $X^2 = 2.4$ ,  $p \leq 0.1225$ ). Larger size classes were predominantly female, with  $\geq 40.0$  cm SCL turtles exhibiting a sex ratio of 2.1F:1.0M ( $n = 236$ ,  $X^2 = 28.5$ ,  $p < 0.0001$ ).

Sex ratio varied significantly across capture years ( $X^2 = 11.92$ ,  $p \leq 0.0077$ ). Females were dominant during 1994 ( $n = 36$ ,  $X^2 = 1.78$ ,  $p \leq 0.1824$ ), 1996 ( $n = 34$ ,  $X^2 = 5.76$ ,  $p \leq 0.0164$ ) and 1997 ( $n = 52$ ,  $X^2 = 9.31$ ,  $p \leq 0.0023$ ), significantly so during the latter two (Fig. 19). Although males dominated during 1993 (1.0F:1.7M) and 1995 (1.0F:1.4M), sex ratios for these years were not significantly different from 1F:1M ( $n = 42$ ,  $X^2 = 3.13$ ,  $p \leq 0.0768$  and  $n = 32$ ,  $X^2 = 1.00$ ,  $p \leq 0.3173$ , respectively).

## Discussion

Gregory and Schmid (in review) reported a sex ratio of 1.8F:1.0M (n = 36) for wild-captured Kemp's ridley sea turtles near Cedar Key, Florida during 1992 (Table 2). Stabenau *et al.* (1996) reported an overall sex ratio of 3.2F:1.0M (n = 144) for *L. kempii* stranded along the upper Texas coast from Sabine Pass to the west end of Matagorda Island during 1986 - 1992. Headstart *L. kempii* in the latter study exhibited a sex ratio of 7.5F:1.0M (n = 17) while wild cohorts were 3.0F:1.0M (n = 127). Danton and Prescott (1988) reported a sex ratio of 1.4F:1.0M (n = 48) for stranded *L. kempii* from Cape Cod, Massachusetts. No mention was made of headstarted individuals. Shaver (1991) reported an overall sex ratio of 1.0F:1.0M (n = 81) for ridleys stranded on south Texas beaches during 1983 - 1989. Forty-two of the observed strandings were headstart Kemp's ridleys (1.6F:1.0M) with the remaining wild cohorts exhibiting a sex ratio of 1.0F:1.8M (n = 39).

Table 2. Overall sex ratios and those for wild and headstarted Kemp's ridley sea turtles reported during this study and other studies.

<b>Sex Ratio</b>	<b>N</b>	<b>Time Frame</b>	<b>Source</b>
1.5F:1.0M overall	251	1993 - 1997	this study
1.3F:1.0M wild	231		
9.0F:1.0M head-start	20		
1.0F:1.0M overall	20	1993	Owens, unpublished report
1.8F:1.0M wild	36	1992	Gregory and Schmid, in review
3.2F:1.0M overall	144	1986 - 1992	Stabenau <i>et al.</i> , 1996
3.0F:1.0M wild	127		
7.5F:1.0M head-start	17		
1.0F:1.0M overall	81	1983 - 1989	Shaver, 1991
1.0F:1.8M wild	39		
1.6F:1.0M head-start	42		
1.4F:1.0M	48	1977 - 1987	Danton and Prescott, 1988

Results of this study suggest an overall sex ratio for Kemp's ridley sea turtles of 1.5F:1.0M (n = 251) versus 1.3F:1.0M (n = 231) for wild cohorts only (Table 2). Headstart *L. kempii* account for a portion of the female bias exhibiting a sex ratio of 9.0F:1.0M (n = 20). This is not surprising as approximately 81% of all headstart ridleys released since 1984, after discovery of the pivotal sex determination temperature for Kemp's ridley, have been females (Stabenau *et al.*, 1996). Their impact will likely decrease as the Head Start Program is no longer in effect (100 - 200 ridleys are still raised each year by NMFS-Galveston for experimental purposes).

A wide-range of sex ratios have been reported by various Kemp's ridley studies and it is possible to derive an array of sex ratios within individual studies (Table 2). Plausible hypotheses to explain differences within and between studies include:

**sexually biased...**

- 1) gonadal decay rates in stranded animals;
- 2) annual sex ratio production;
- 3) stranding and/or capture rates;
- 4) movement or migration patterns;
- 5) geographic distribution; or
- 6) survival/mortality rates.

Evidence of sexually biased gonadal decay rates (Hypothesis 1) among Kemp's ridleys has been provided by Owens (unpublished report) who examined gonads of 44 1-5 kg carcasses stranded in the Grand Isle, Louisiana area during Spring 1993. Owens noted that testes were more compact and solid while immature ovaries were very thin and tissue-like, possibly making it more difficult to distinguish ovaries in highly decomposed individuals. This difficulty in distinguishing ovarian tissue could presumably cause an erroneous male bias in analysis of stranded animals because a greater proportion of males would be identifiable. However, two studies cited in Table 2

suggest a female bias in stranded *L. kempii* (Danton and Prescott, 1988; Stabenau *et al.*, 1996). Owens' examination occurred a year after the stranding event and involved frozen carcasses previously necropsied for other evaluations. Stabenau *et al.* (1996), on the other hand, only necropsied relatively fresh carcasses.

Evidence to support annual variation in hatchling sex ratio production is lacking (Hypothesis 2). The fact that most *L. kempii* nests (>90%) have been transplanted to corrals since 1978 (Márquez, 1994) has resulted in eggs from a given arribada being subjected to similar environmental conditions which dictate sex determination during incubation. However, seasonal and annual variation in factors such as temperature and rainfall should be sufficient to insure production of both sexes. While annually sampled sex ratios of juvenile to adult *L. kempii* appear to vary considerably (Fig. 19), it is unlikely that this is a function of hatchling sex ratio production, but instead related to Kemp's ridley behavior and/or distribution dynamics. In any case, four studies mentioned previously (Table 2) covered a period of at least four years. While longer studies are desirable, this should help mitigate confounding annual effects.

Sex-specific migration patterns, distribution and mortality rates (Hypotheses 3 - 6), are related in that they are contingent upon *L. kempii* movement and/or behavior. It is unclear whether males and females utilize similar habitats or ranges at each life history stage or whether adults of each sex follow the same reproductive migratory path to the nesting beach. Such differences could have profound effects upon exposure to various mortality factors, subsequent stranding rates (percentage of dead animals that actually end up on the shore), and exposure to near-shore capture efforts, such as this study.

Wibbels *et al.* (1987, 1990) suggested that increased circulating testosterone (T) in adult male loggerhead sea turtles may affect or coincide with migration and other reproductive events including spermatogenesis, courtship, and mating. A similar peak coinciding with Spring interpond breeding migration has been suggested (Gibbons,

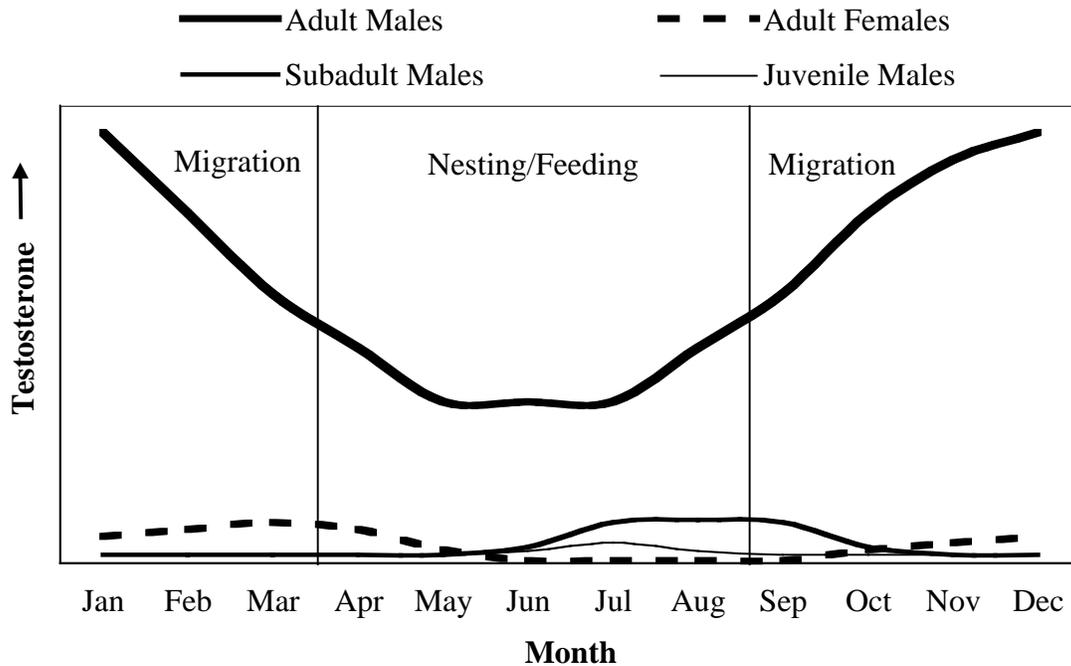


Figure 20. Generalized testosterone model at each life history stage of male and female Kemp's ridley sea turtles. Adult responses adapted from Rostal (1991) and Morris (1982).

1968) for plasma T levels in male red-ear turtles (*Chrysemys picta*). Male slider turtles (*Pseudemys scripta*) and Concho water snakes (*Nerodia harteri paucimaculata*) have been reported to emigrate more frequently and farther than their female counterparts, while younger *P. scripta* males appear to be more sedentary than larger males (Parker, 1984; Whiting *et al.*, 1997).

Figure 20 presents a hypothetical testosterone model for *L. kempii* by life history stage adapted from data collected during this study and from Morris (1982), Rostal (1991) and Rostal *et al.* (1997). In general, the model suggests that adult male *L. kempii* maintain a relatively high level of testosterone year round, which builds to a peak coinciding with mating migration and then slowly decreases to baseline again. Adult females exhibit a similar response, with testosterone levels peaking at or near the onset

of migration, and then slowly decreasing to baseline during the nesting season. Seasonal analyses suggest plasma T in juvenile and subadult male *L. kempii* in the Gulf of Mexico are characterized by a slight increase during June through September (Coyne and Landry, 2000). Juvenile and subadult females are expected to exhibit little variation relative to their male counterparts as their baseline plasma T levels are considerably lower.

While plasma T baseline values and ranges typical of each sex/life-stage of *L. kempii* are not well understood, the following values have been reported in the literature and observed during this study. Maximum plasma T detected during this study (2063 pg/ml) was from a confirmed headstart male 59.7-cm SCL. Rostal *et al.* (1997) reported a maximum plasma T of 219 pg/ml from a nesting female at Rancho Nuevo, Mexico. Maximum plasma T observed in non-nesting, 60+ cm SCL females captured during this study was 5.07 pg/ml (n = 6). Plasma T in confirmed male juveniles (circa 20 - 40 cm SCL) ranged from 18 - 213 pg/ml ( $\bar{x} = 51.5 \pm 11.2$ ) and 68 - 2063 ( $\bar{x} = 394.6 \pm 172.5$ ) in subadults (circa 40 - 60 cm SCL). Juvenile and subadult females exhibited ranges of 0.2 - 12.0 ( $\bar{x} = 5.3 \pm 0.9$ ) and 0.2 - 8.2 ( $\bar{x} = 4.6 \pm 0.6$ ), respectively.

Elevated plasma T associated with the nesting season (Morris, 1982; Rostal 1991; Rostal *et al.*, 1997) may be partially responsible for initiating migration in females. Satellite telemetry (Renaud *et al.*, 1996) from an adult female captured at Calcasieu Pass (8/11/95) and later observed nesting at Rancho Nuevo (4/23/96 and 5/19/96) supports this supposition as she began a concerted southerly track at approximately the same time (December) plasma T is suggested to begin rising (Rostal, 1991). Owens (1997) suggests that reproductive migration in female sea turtles coincides with a peak in annual plasma T concentration and that adult males experience an earlier peak and thus begin migration earlier. These observations are speculative, as direct experimentation linking testosterone to migration behavior has not been completed.

Despite a small sample size ( $n = 11$ ), previous analysis of plasma T data used in the current study suggests an exponential increase in circulating T with size in subadult male Kemp's ridleys (Coyne and Landry, 2000). The apparent plasma T increase in subadult males may result in a larger homerange and/or alteration of habitat utilization, similar to the proposed elevated plasma T initiation of migration in adults. This hypothesis is supported by radio and satellite telemetry suggesting that larger *L. kempii* utilize deeper waters and larger home ranges (Renaud *et al.*, 1995).

Plasma T clearly plays a role in seasonal reproduction in both male and female Kemp's ridleys and may play a behavioral role in triggering female receptivity and onset of mating (Rostal, 1991). However, while plasma T may influence Kemp's ridley behavior and movement patterns, a myriad of factors are probably involved, including temperature, season, reproductive status, and prey availability.

Finally, there is the possibility that one or more of the aforementioned studies (Table 2) does not accurately reflect Kemp's ridley population demographics. Size distribution (21.8 - 62.1 cm SCL) of turtles sexed during this study was representative of the entire capture lot (Fig. 14). It is unclear whether the same is true of previous sex ratio studies utilizing stranded Kemp's ridleys (Danton and Prescott, 1988; Shaver, 1991; Stabenau *et al.*, 1996). Size distribution of wild Kemp's ridleys captured at Sabine Pass during this study correlate well (Pearson product-moment = 0.9220) with wild ridleys stranded during the same period (1993 - 1996) in NMFS statistical subarea 18 (Sabine Pass through Galveston County, Texas) (Fig. 21). However, these stranding data include smaller cohorts not encountered in netting operations and a greater proportion of subadult (40 to 60 cm SCL) and adult turtles (greater than 60 cm SCL). Size distribution of stranded Kemp's ridleys indicated in figure 21 is consistent with those presented by Shaver (1991) and Stabenau *et al.* (1996).

While stranding dynamics are beyond the scope of this study, the proposed

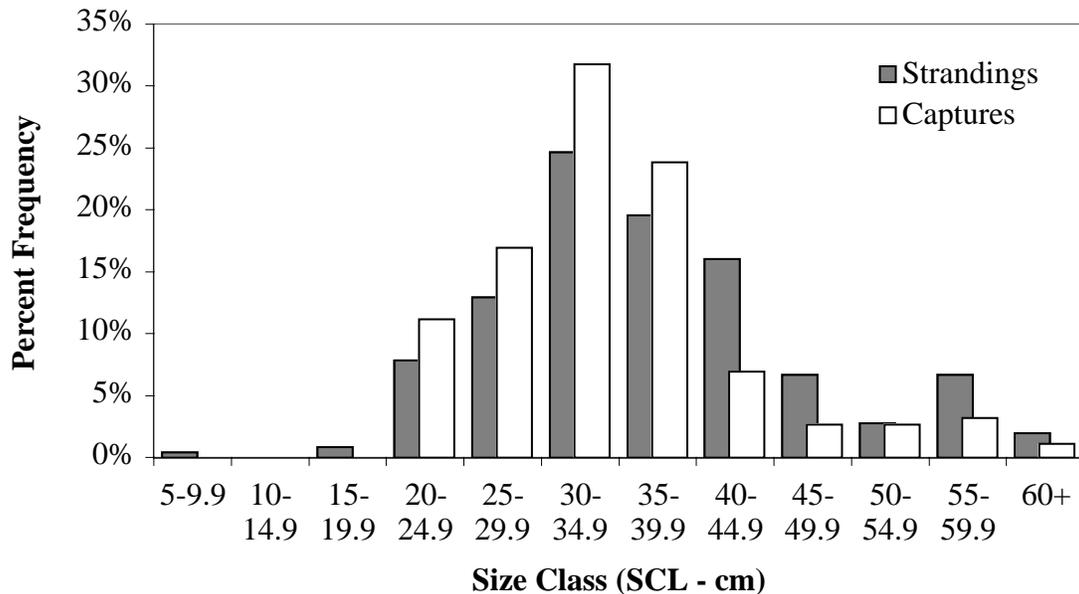


Figure 21. Size distribution of wild Kemp's ridley sea turtles captured at Sabine Pass ( $n = 189$ ) versus those found stranded in NMFS Statistical Zone 18 ( $n = 256$ ) during the same period (1993-1996). Pearson product-moment correlation = 0.9220.

testosterone model (Fig. 20) for Kemp's ridley sea turtles may well explain observed differences in stranding and capture sex results. Shaver (1991) and Stabenau *et al.* (1996) reported similar size distribution patterns of juvenile Kemp's ridleys suspected to inhabit shallow, nearshore waters, where they become more susceptible to capture by nearshore netting operations and post-mortem stranding. Stranded post-juvenile ridleys sexed by Stabenau *et al.* (1996) exhibited an increasing female bias with size. Perhaps increased plasma T in subadult (pubescent) males is, in part, responsible for this observation. Larger males move farther from shore and/or maintain larger homeranges thereby: 1) decreasing the chance that a dead animal will reach shore; 2) enhancing the chance the carcass will reach shore in a deteriorated state making sex determination more difficult; 3) increasing the likelihood of being exposed to different mortality pressures; or 4) placing them out of range of near-shore capture efforts. Females, which

only exhibit increased plasma T associated with the breeding season, may otherwise remain closer to shore and thus are more susceptible to nearshore capture, mortality and/or stranding.

Similarly, the majority of female bias observed by Stabenau *et al.* (1996) also applies to larger size classes, particularly ridleys greater than 60 cm SCL. A review of only wild juvenile and subadult turtles (20 to 60 cm) examined by Stabenau *et al.* yielded a sex ratio of approximately 2.3F:1.0M, closer to but still considerably greater than the 1.3F:1.0M (n = 190) observed during this study. The disparity between results from these two studies is even greater for  $\leq 40$  cm (1.2F:1.0M, n = 149) and  $\leq 30$  cm SCL (1.0F:1.0M, n = 43) juveniles sexed in the current study.

This model of testosterone-behavior interaction may help explain increasing female bias with size observed during this study and the exclusive capture of females among Kemp's ridleys greater than 60 cm SCL. Of these eight larger turtles, six were verified as females and two others exhibited plasma T levels consistent with female Kemp's ridleys (5.4 and 10.3 pg/ml, respectively). The largest confirmed or predicted male captured in this study was 59.1 cm SCL. Although more difficult to verify, an alternative explanation is that maturing males suffer greater mortality and thus exhibit lower abundance at maturity than do adult females. However, the fact that both stranding and capture data exhibit similar sex ratio patterns does not support such a case.

If one assumes that male *L. kempii* are not subject to differential mortality then the 1:1 sex ratio observed in juveniles most closely represents the primary sex ratio of the population. Implicit in this assumption is that subadult and adult males are encountered less frequently than their female counterparts because of differences in behavior and/or habitat use. However, sex ratio variations reported across size classes and in stranded animals cannot be dismissed, particularly considering data presented by Stabenau *et al.* (1996).

### CHAPTER III

#### POPULATION MODEL

Demographic models allow one to simulate a population's response to various factors and assess the relative importance of each model variable to the final product. Simulation modeling as a management tool has provided the impetus to model animal populations either to assess their status, validate potential management scenarios for threatened populations, or develop harvest quotas for commercially valuable species (Heppel and Crowder, 1994, 1996; Heppel *et al.*, 1995; Heppel, 1998; TEWG, 1998). These population models are generally constructed from life table data, of which essential parameters continue to elude sea turtle investigators (Bustard, 1979)

Until recently only portions of sea turtle population models had appeared in the literature (Richardson and Richardson, 1982). For example, Hughes (1974) used annual egg production, egg survival, and observed recruitment to an adult nesting loggerhead population to estimate juvenile survival rates. Various investigators have used remigration-interval frequency and seasonal population counts to estimate number of nesting females (Carr *et al.*, 1978; Márquez, 1994; TEWG, 1998). Bustard and Tognetti (1969) developed a model for density-dependent population regulation through the mechanism of intraspecific nest destruction.

The need to assess current and proposed management efforts has resulted in recent attempts to develop comprehensive models for threatened and endangered sea turtle populations. Models developed for loggerhead sea turtles have suggested that it may be more valuable to protect older cohorts at sea, for example through the use of turtle excluder devices (TEDs), than those on the nesting beach (Crouse *et al.*, 1987; Heppel *et al.*, 1996a). Similar models have been developed to assess the status of loggerhead and Kemp's ridley populations (TEWG, 1998). However, the paucity of data prohibits these models from providing reliable quantitative analyses of important life-

history parameters, like survivorship and age-at-maturity.

Chaloupka and Limpus (1996) presented preliminary results of robust statistical models based upon long-term studies and relatively large datasets. These models represent the first opportunity to overcome shortcomings of previous modeling efforts. For the most part, few data have been available for sea turtles from the time they leave the nesting beach as hatchlings until adult females return to nest. Uncertainty surrounding age-at-maturity and the fact that survivorship is likely highly variable has made it difficult to calculate survivorship for interim life-history stages or to assess the impact of management efforts.

A great deal is known about Kemp's ridley nesting beach behavior and reproductive parameters, primarily as a result of the ongoing 20+ year joint U.S.-Mexico conservation program at Rancho Nuevo, Mexico. However, attempts to model the Kemp's ridley population still lack reliable estimates for those critical variables associated with at-sea life-history stages (Márquez, 1994; Heppel *et al.*, 1996b; TEWG, 1998). This chapter focuses on the impact of various sex ratio distributions upon Kemp's ridley population demographics by developing a population model based upon the work of other investigators and through insights gained from at-sea capture of Kemp's ridleys during sex ratio investigations.

The effect of sex ratio upon population dynamics in sea turtles is poorly understood. It has been suggested that managers alter hatchling sex ratios through artificial incubation or application of estrogen in order to influence population sex ratios (Vogt, 1994). This suggestion relies on the assumption that female turtles are more important than males because one male can inseminate many females. However, caution should be used until the potential demographic and ecologic consequences of manipulating sex ratios in turtle populations are better understood (Lovich, 1996).

Multiple paternity has been documented in several turtle species including

snapping (*Chelydra serpentina*) and loggerhead turtles (Harry and Briscoe, 1988; Galbraith *et al.*, 1989, 1993; Bollmer *et al.*, 1999). In addition, Gist and Jones (1989) and Palmer *et al.* (1998) documented several turtles species possessing the ability to store sperm. Together these characteristics raise interesting questions about their adaptive advantages (Lovich, 1996). What effect does multiple paternity and the ability to store sperm have on population size, offspring viability, or fecundity?

Sugg and Chesser (1994) modeled gene correlations in populations with different mating strategies. They suggest that multiple paternity increases effective population size over that expected from polygyny and monogamy by maintaining genetic variation. Madsen *et al.* (1992) suggested that multiple copulations by female adders (*Vipera berus*) might enhance offspring viability, either because of inadequate quantities of sperm from a single mating, additional nutrients derived from the seminal fluid, or some genetic advantage. Mrosovsky and Godfrey (1995), citing Chan (1991), suggested that poor hatch rates of leatherback sea turtles (*Dermochelys coriacea*) in Malaysia may be attributable to an insufficient number of males to fertilize clutches. Additionally, Wood and Wood (1980), observing captive green sea turtles, reported the percentage of nesting females was directly correlated with mating duration, also noting that mating duration was partially a function of the number of males that copulated with a female.

These potential adaptive advantages of multiple paternity and sperm storage are only effective when the number of reproductive males is sufficient to facilitate multiple insemination of reproductive females. Taking these influences into account, the model developed in this chapter will be used to assess affects of various sex ratios upon the Kemp's ridley population under the following assumptions:

- 1) sex ratio does not influence fecundity or offspring viability; and
- 2) male bias increases fecundity or offspring viability.

The effect of these assumptions will be assessed by altering clutch frequency and size in relation to sex ratio (see next section - Model Formulation and Quantification).

Changing hatch success in relation to sex ratio was not considered as this variable is extremely biased by the practice of protecting nests on the nesting beach. Fecundity effects will be analyzed in relation to sex ratio as discussed in Chapter II. The Kemp's ridley population sex ratio was considered to be 1F:1M for the baseline model.

Additional sex ratio scenarios will be tested to assess sensitivity of the model to sex ratio changes and potential impact upon the population including: 3F:1M, 2F:1M, 1F:2M and 1F:3M.

### **Model Formulation and Quantification**

The strategy used to develop the present model was to sacrifice precision to realism and generality (Levins, 1966). In other words, the model does not attempt to predict the population exactly, only the population's response to changing parameters built into the model. Primary concern with qualitative rather than quantitative results permits a flexible, graphical model to be developed which generally assumes that functions are increasing or decreasing, greater or lesser than some value, instead of specifying the mathematical form of an equation. The litany of variables needed for a robust statistical model can be reduced to manageable proportions by abstracting many functions into a reduced number of higher-level functions. For example, it is not necessary to know how many eggs are non-viable and how many succumb to predation or inundation if you can estimate the proportion of eggs laid that develop into hatchlings.

The basic process to be modeled was annual size of the Kemp's ridley population. Number of nests was used as an indicator of the population as a whole because there is currently no way to estimate total population size or even that of a given life-history stage. Comprehensive monitoring of the Rancho Nuevo nesting beach since

1978 makes annual number of nests one of the best indices of population status for Kemp's ridley, assuming that number of nests laid each year is an appropriate index (TEWG, 1998).

The Kemp's ridley life cycle was grossly simulated to estimate annual number of nests (Fig. 22; Table 3). Key components of the life cycle were evaluated including variables related to: 1) reproduction; 2) growth/age; and 3) mortality. Lacking data and understanding regarding *L. kempii* growth and mortality rates render components 2 and 3 the most difficult to solve.

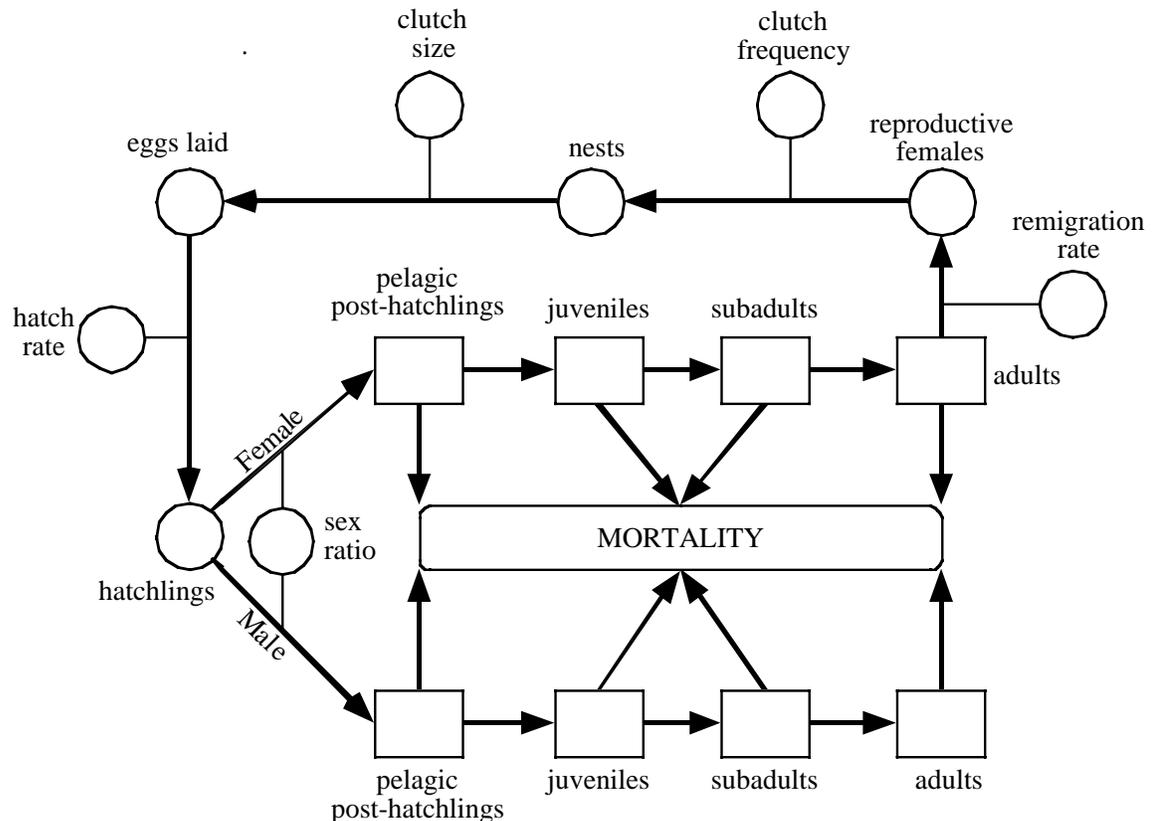


Figure 22. Conceptual representation of the Kemp's ridley demographic model.

Table 3. Definition of variables used in Kemp's ridley population model.

<b>Variable</b>	<b>Definition</b>
<i>Mortality</i>	Annual life-stage specific mortality rate.
<i>Post-hatchling</i>	Pool of animals, presumably in the pelagic environment, encompassing the time when <i>hatchlings</i> leave the beach until becoming <i>juvenile</i> . Typically referred to as "lost-year".
<i>Juvenile</i>	Post-pelagic pool of animals of a size and/or age at which little or no sexual development is occurring.
<i>Subadult</i>	Pool of animals of a size and/or age at which sexual development is occurring.
<i>Adult</i>	Pool of animals that have reached sexual maturity.
<i>Sex Ratio</i>	Proportion of female and male turtles as determined by this study.
<i>Remigration Rate</i>	Proportion of <i>adult</i> females that nest in a given year.
<i>Reproductive Females</i>	Number of <i>adult</i> females that nest in a given year as determined by <i>remigration rate</i> .
<i>Clutch Frequency</i>	Mean number of clutches per season per <i>reproductive female</i> .
<i>Nests</i>	Number of nests deposited by all <i>reproductive females</i> as a function of <i>clutch frequency</i> .
<i>Clutch Size</i>	Mean number of <i>eggs laid</i> per clutch.
<i>Eggs Laid</i>	Number of eggs laid each nesting season as a function of <i>nests</i> and <i>clutch size</i> .
<i>Hatch Rate</i>	Proportion of hatchlings from <i>eggs laid</i> that successfully leave the nesting beach.
<i>Hatchlings</i>	Total number of <i>hatchlings</i> that survive from <i>eggs laid</i> as a function of <i>hatch rate</i> .

## Reproduction

**Remigration Rate:** Adult female Kemp's ridleys do not necessarily nest every year.

Remigration rate is a measure of the mean inter-annual nesting interval or proportion of mature females that nest each year. Inter-annual renesting frequency evaluated at the Rancho Nuevo, Mexico nesting beach using tag and recapture data indicates that 58% of adult females nest every year, 29% every two years, and 13% every three years, resulting in an annual remigration rate of 0.768 (Márquez, 1994). The Turtle Expert Working Group (TEWG, 1998) estimated that each female nests every 2.05 years, a remigration rate of 0.575, based upon Márquez's (1994) published mark-recapture data and more recent unpublished data (25% every year, 60% every two years, 15% every three years, and 10% every four years). However, the source of these data are unclear and may underestimate remigration rate.

Loss of monel-style flipper tags has been a problem for most sea turtle species, making it difficult to identify individuals during subsequent observations (Mrosovsky, 1976). Although it is possible that more reliable PIT tags were used in this assessment, Burchfield *et al.* (1995, 1996, 1997, 1998) reported that only 18.7, 97.5, 79.6, and 78% of observed turtles were scanned for PIT tags during 1995 - 98, respectively. Nesting events are missed because of insufficient resources to completely survey the nesting beach temporally and spatially. The inability to confirm a nesting female's identity during subsequent nesting seasons can result in an underestimate of remigration rate. Consequently, the larger value of 0.768 (Márquez, 1994) was used in the model to represent proportion of mature females nesting each year.

**Clutch Frequency:** Mature Kemp's ridley females may nest multiple times in a given season. Clutch frequency is a measure of the mean number of clutches laid by nesting females each season. Márquez (1994) reported a clutch frequency, or fecundity index, of 1.3. Another study reported clutch frequency to be 1.4 - 1.55 (Márquez, 1990).

Pritchard (1990) reported a clutch frequency of 2.3 for the 1989 nesting season based on the statistical probability of observing first-, second- and third-time nesters. All of these studies relied on flipper tags to identify returning females and are subject to the same tag loss and missed nesting event problems associated with calculating remigration rate.

The TEWG (1998) cited unpublished data (Márquez) from a recent PIT tag study suggesting a rate of 1.8 nests/female/season. PIT tags do not present the same retention problem observed with flipper tag use, but can be difficult to locate. Rostal (1991) and Rostal *et al.* (1997) suggested that Kemp's ridley sea turtles were capable of 3.075 clutches per season based upon an ultrasound study of nesting females. This study presents the most reliable quantitative data because missed nesting events can be accounted for through determination of ovarian status (gravid versus depleted follicles) when the turtle is observed. These data were supported by a concurrent testosterone titer study indicating that testosterone declined through the nesting season as the source of testosterone is removed (ovulation of the follicles). It should be noted that this study included ultrasonography data from only one nesting year (1990) and a limited sample size (56 ultrasound scans of 50 turtles). Nevertheless, the resulting 3 clutches/female/season was used in the model as the most reliable estimate to date.

To assess potential effects of changing fecundity in relation to sex ratio clutch frequency was adjusted proportionally to the difference in percent adult female from 50% (Fig. 23). For example, a 2F:1M adult sex ratio equals a percent female of 66% and, thus, clutch frequency would be decreased by 16% ( $50 - 66 = -16$ ). If, on the other hand, sex ratio is 1F:3M, then percent female would be 25% and clutch frequency would be increased by 25% ( $50 - 25 = 25$ ). This fecundity adjustment is arbitrary and not based on qualitative or quantitative analyses.

**Clutch Size:** Clutch size represents the mean number of eggs laid in each clutch. A mean of 100.8 eggs per clutch was reported for the 14 year period after the U.S.-Mexico

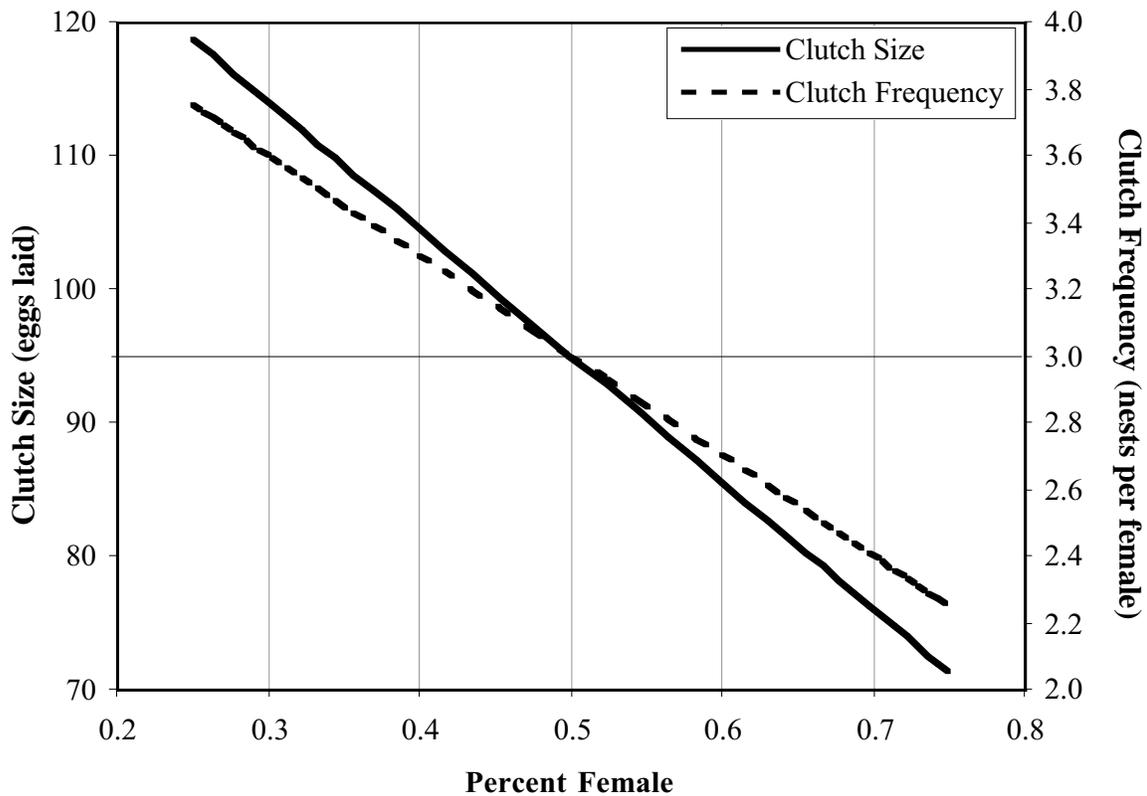


Figure 23. Arbitrarily assigned fecundity effect. Clutch size and clutch frequency change proportionally with increasing male bias.

Kemp's ridley conservation project began in 1978 (USFWS/NMFS, 1992). Márquez (1994) reported a mean clutch size of 104 during 1966 - 1992, but also suggested that the number of eggs per nest had been declining. Analysis of these and more recent data (Burchfield *et al.*, 1995, 1996, 1997, 1998) supports this suggestion indicating a significantly decreasing trend ( $F = 70.46$ ,  $p \leq 0.0001$ ) in mean number of eggs per clutch since 1966 (Fig. 24).

Decreasing clutch size may be the result of an aging adult female population with little or no recruitment during the 1970's and early 1980's. There is evidence to suggest a recent influx of new recruits since 1988 related to increases in number of nests reported (Fig. 25). Direct observation of neophyte nesters at nesting beaches also

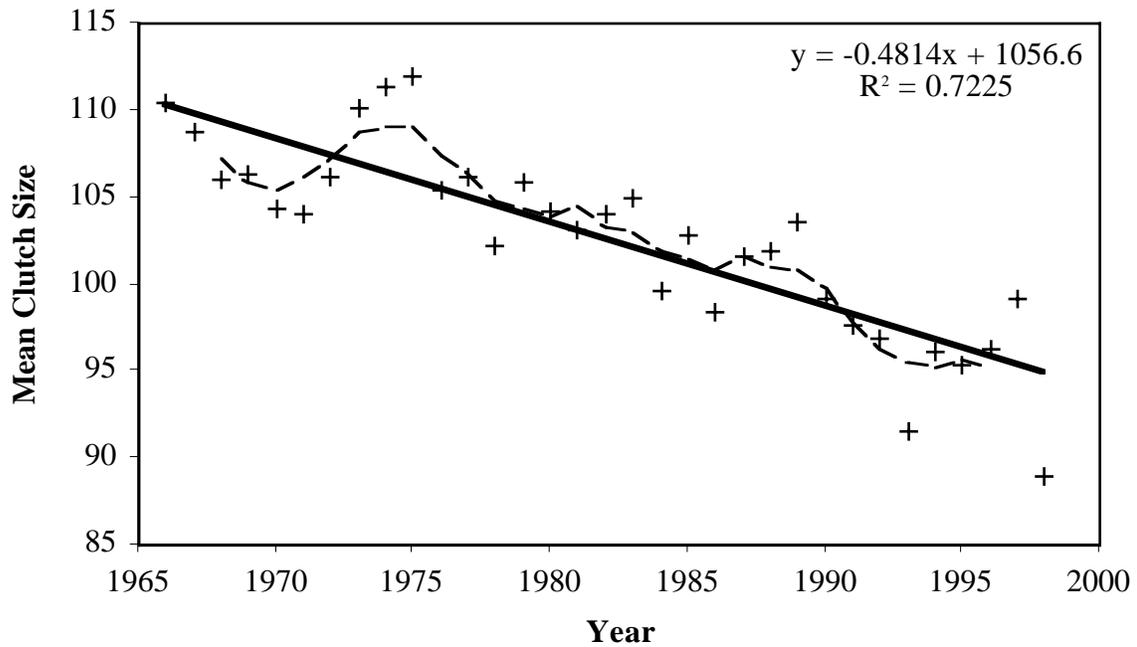


Figure 24. Mean Kemp's ridley clutch size reported at Rancho Nuevo, Mexico during 1966 - 1998 (Márquez, 1994; Burchfield *et al.*, 1995, 1996, 1997, 1998; J. Peña, Pers. comm.). Mean number of eggs per clutch has decreased significantly since 1966 ( $n = 30$ ,  $p \leq 0.0001$ ). Dashed line is a 5-year running average and solid line is fit-line.

indicates an increasing number of first-time nesters, with proportion of neophytes increasing from 13 to 56% during 1981 - 1989 and estimates of greater than 40% in recent years (Márquez, 1994). In addition, Burchfield *et al.* (1994, 1995, 1996, 1997, 1998) estimated an increasing number of observed neophyte nesters during 1994 - 98 (271, 420, 473, 516, and 548, respectively).

Rostal (1991) and Rostal *et al.* (1997) noted a weak positive correlation between size of nesting Kemp's ridleys and clutch size. Similar observations have been made for the olive ridley and green sea turtles (Pritchard, 1969; Bustard, 1973). An increasing number of neophyte nesters, producing fewer eggs than their more experienced counterparts, may partially account for observed decreases in average clutch

size since 1966. Clutch size will likely continue to fluctuate until the adult female population reaches a stable age distribution. Mean clutch size of 95 eggs/clutch, observed during the last five years for which data are available (1994-1998), was used in the model to estimate clutch size.

As with clutch frequency, the potential effects of changing fecundity in relation to sex ratio will be assessed by adjusting clutch size by a percentage equal to the change in percent adult female from 50% (Fig. 24).

**Hatch Rate:** Since the inception of the U.S.-Mexico conservation project in 1978, 65% of all collected eggs at the Rancho Nuevo nesting beach have been released as hatchlings (Burchfield *et al.*, 1998). Relocation of almost all *L. kempii* nests to protected corrals is responsible for this relatively high level of hatchling survival. In addition, hatchling survival rates have increased annually since the project first began, with all but one year since 1984 (1998) exhibiting rates above 60%. Mean hatch rate during that time was

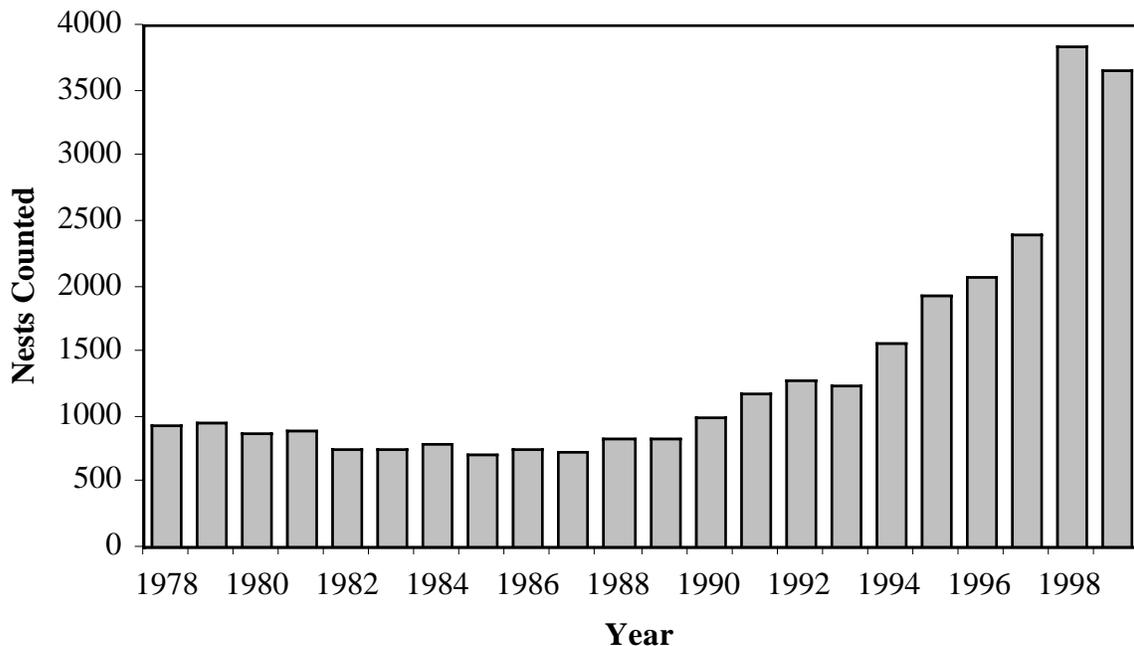


Figure 25. Number of Kemp's ridley nests counted each season during the binational U.S.-Mexico conservation project at Rancho Nuevo, Mexico since 1978.

70.5% and has ranged from 53.1 to 79.1%. Known hatch rates were used in the model where available (1978 - 1998). The 1984 - 1998 mean hatch rate (0.705) was used for all subsequent years.

### **Growth and Age**

Although skeletochronological techniques have been used to estimate the age of stranded carcasses (Zug, 1990), there is no reliable method for determining age in sea turtles. In general, skeletochronological data from sea turtles yield age-estimates, not actual ages, used in predicting the population's growth pattern and "average" age at maturity (Zug *et al.*, 1997). Some weaknesses of this method include the resorption of annular rings making it difficult to assess age in older turtles and a lack of age validation to adjust for this loss (Zug *et al.*, 1986, 1997; Zug 1990, 1991; Chaloupka and Musick, 1997; Chaloupka and Zug, 1997). In addition, efforts employing this method on *L. kempii* have primarily included coastal Atlantic cohorts which may exhibit a slower growth rate than Gulf of Mexico conspecifics (Caillouet *et al.*, 1995).

Most studies estimating age in sea turtles have relied on recapture growth data. Unfortunately, obtaining comprehensive growth data for sea turtles, particularly *L. kempii*, has proven problematic because of their marine existence and the difficulty in recapturing individual turtles throughout their size range.

The method utilized most often to estimate growth in sea turtles, including green, loggerhead, and, more recently Kemp's ridley, is to fit growth interval data to logistic, Gompertz, or von Bertalanffy growth curve equations (Frazer and Ehrhardt, 1985; Frazer and Ladner, 1986; Zug, 1990; Caillouet *et al.*, 1995). The von Bertalanffy equation has provided the best fit (Frazer and Ehrhardt, 1985; Caillaouet *et al.*, 1995; Schmid and Witzell, 1997; Zug *et al.*, 1997) and has been utilized in five different studies of Kemp's ridleys, including this one (Table 4). The von Bertalanffy

Table 4. Von Bertalanffy growth equation parameters, size-at-maturity, and estimated age-at-maturity for Kemp's ridley sea turtles from various studies ( $k$  = intrinsic growth rate;  $a$  = asymptotic length;  $N$  = number of turtles).

$k$	$a$ (SCL-cm)	Size-at-Maturity (SCL-cm)	Age-at-Maturity (Years)	$N$	Data Source
0.317	62.27	60	10	114	tag returns stranded head-starts (Caillouet <i>et al.</i> , 1996)
0.2	63.95	59.5	11-12	44	skeletochronology age estimate (Zug, 1990)
0.259	78	not given	7	?	nesting female returns (Marquez, 1972, 1994)
0.1292	80	(56.0,60,64.2)	(8-9,10-11,12-13)	36	recaptured wild turtles in Florida (Schmid and Witzell, 1997)
0.259	69.36	62.5	9†	61	recaptured wild and head-start turtles and literature review, this study

† one year added (10 years total) for model to compensate for lacking “lost-year” data.

growth equation is generally expressed as:

$$L_t = a (1 - be^{-kt})$$

where  $L_t$  is the length at recapture,  $a$  is the asymptotic length,  $b$  is a parameter related to size at hatching,  $k$  is the intrinsic growth rate, and  $t$  is age. The von Bertalanffy growth interval equation was modified by Fabens (1965) in order to estimate growth parameters of animals of unknown age based on recapture data. The Fabens growth equation is:

$$L_r = a - (a - L_c)e^{-kd}$$

where  $L_r$  is the length at recapture,  $a$  is asymptotic length,  $L_c$  is the length at first capture,  $k$  is the intrinsic growth rate, and  $d$  is the time interval between captures. The Fabens equation does not contain an estimate of  $b$ , which is necessary to complete the von Bertalanffy model. This parameter was estimated using the von Bertalanffy equation simplified to time zero ( $t = 0$ ):

$$b = 1 - L_0/a$$

where  $L_0$  is the mean hatchling carapace length of 4.4 cm (Márquez, 1994).

A major limitation of the von Bertalanffy equation is that it is monophasic. It has been suggested that *L. kempii* exhibits a polyphasic growth function, going through more than one growth spurt during their lifetime (Chaloupka and Musick, 1997; Chaloupka and Zug, 1997).

Recapture data for this study ( $n = 34$ ) included six recaptures of wild turtles, two stranding tag returns of turtles initially wild captured, and 26 captures of released headstart Kemp's ridleys (Appendix A). Use of recaptured headstart Kemp's ridleys in this analysis was considered valid since previous studies have shown they exhibit diet and behavior patterns similar to those of wild cohorts (Shaver, 1991; Werner, 1994; Landry *et al.*, 1996). Additional data ( $n = 27$ ) from other studies were included to supplement missing size ranges, primarily for ridleys greater than 60 cm SCL. These included length data obtained from tagged renesters at Rancho Nuevo, Mexico and other tagged nesters found stranded (Chavez and Kaufman, 1974; Márquez, 1994). Data obtained (McVey and Wibbels, 1984) from recapture or stranding of released headstart individuals were also included. Outliers excluded from the analysis included 31 datapoints exhibiting negative growth or positive growth greater than 20 cm/yr. In addition, individuals at-large for less than 0.8 years were excluded to minimize seasonal growth effects. Time at-large for the remaining individuals ranged from 0.8 - 9.8 years ( $\bar{x} = 1.9$  yrs). Growth rates ranged from 0 - 17.5 cm/yr ( $\bar{x} = 8.0$  cm/yr).

The fitted von Bertalanffy growth equation for these data was:

$$b = 1 - 4.4/69.4 = 0.945$$

$$L_t = 69.4 (1 - 0.945 e^{-0.259t})$$

Asymptotic length ( $a$ ) and intrinsic growth rate ( $k$ ) for 95% confidence intervals were 66.4 and 72.6 cm SCL and 0.2294 and 0.2908, respectively (Fig. 26).

**Size-at-maturity:** Calculation of age at sexual maturity from growth curves is problematic, even with the most extensive data sets, because it is difficult to determine

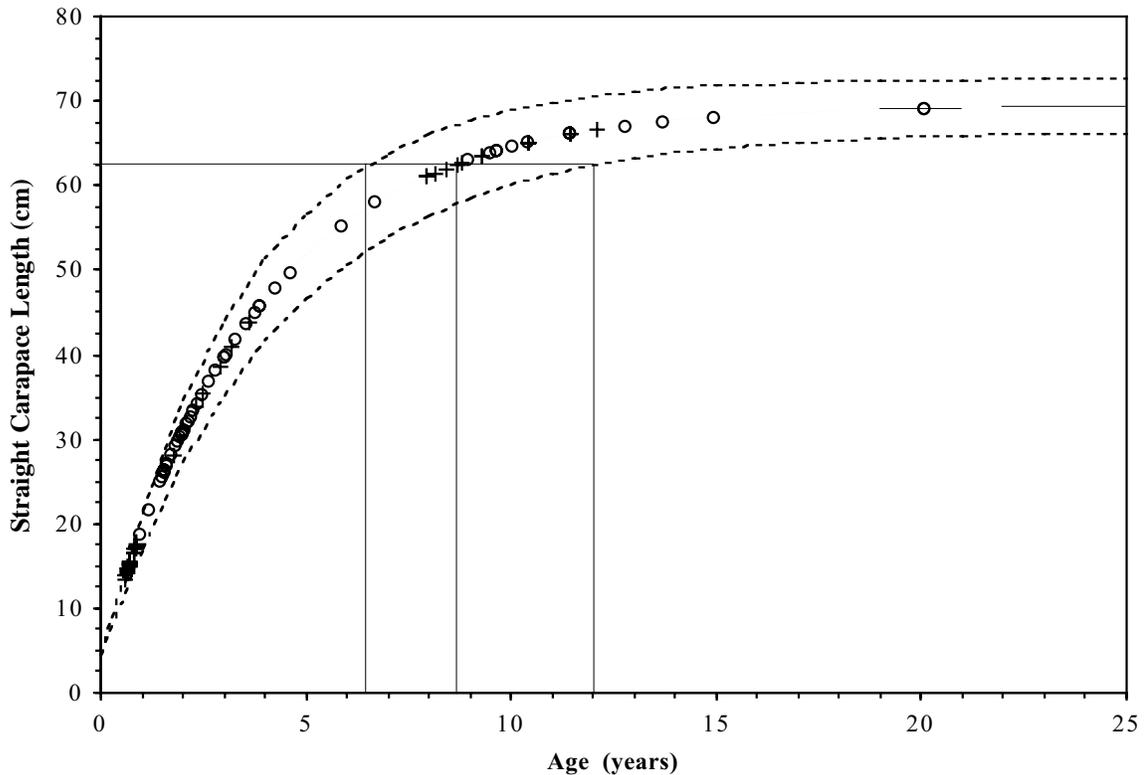


Figure 26. Estimated age-at-maturity for Kemp's ridley sea turtles based upon von Bertalanffy growth interval equation and size-at-maturity estimate of 62.5 cm SCL [  $L_t = 69.36(1 - 0.945e^{-0.259t})$  ]. Dashed lines represent 95% confidence intervals. Estimated age for initial and recapture sizes of included datapoints are shown as + and o, respectively.

size at sexual maturity for marine turtles (Limpus and Chaloupka, 1997). For *L. kempii*, Caillouet *et al.* (1995) assumed a mean size at sexual maturity of 60 cm SCL, and Zug (1990) estimated a minimum and mean of 60 and 65 cm SCL, respectively, based upon minimum and mean size of nesters. Schmid and Witzell (1997) used summary data provided by Burchfield *et al.* (1988) to estimate the size range of nesting adult females as 56.0 - 72.5 cm SCL (n = 468), with a mean of 64.2 cm. These authors used minimum and mean observed nesting length to estimate age-at-maturity. However, using mean size of all nesters is inappropriate because mean size-at-maturity should represent mean

neophyte nester size, not the mean size of all nesters. While growth is probably negligible upon reaching sexual maturity (Limpus and Chaloupka, 1997), the mean size of all nesting turtles is likely slightly higher than that for neophyte nesters alone (Caillouet *et al.*, 1995; Chaloupka and Zug, 1997; Schmid and Witzell, 1997). For example, Limpus (1990) reported that female loggerheads at the Great Barrier Reef recruit at a size slightly smaller than the average breeding size of the entire population. Slower adult growth could have a significant effect if size-at-maturity is over-estimated even slightly, resulting in a greatly inflated age-at-maturity estimate. Whatever the actual value of mean size-at-maturity, it should be less than mean nesting size (64.2 cm SCL). Therefore, a median value (62.5 cm SCL) between the two values most often used (60 and 65 cm SCL) was assumed to represent size-at-maturity for this model.

**Age-at-maturity:** Use of von Bertalanffy growth curves for *L. kempii* has resulted in previous age at sexual maturity estimates of 8-13 years (Zug, 1990; Caillouet *et al.*, 1996; Schmid and Witzell, 1997). However, care should be taken when using these estimates, and the von Bertalanffy equation in general, that estimated equation parameters reflect reality (Table 4).

Schmid and Witzell (1997) suggested that the asymptotic length for *L. kempii* should be close to the mean maximum size observed (72.5 cm SCL). However, using the von Bertalanffy equation with mark-recapture data from turtles captured in Florida, these authors derived an asymptotic length of 80 cm SCL. The TEWG (1998) identified 75 cm SCL (95% of the size of the largest nester) as the asymptotic length for *L. kempii*. Caillouet *et al.* (1996) derived an asymptotic length of 62.27 cm SCL from mark-recapture data on headstarted turtles. Zug (1990), using age estimates for *L. kempii* from skeletochronology, derived an asymptotic length of 63.95 cm SCL while Márquez (1972, 1994) calculated an asymptotic length of 78 cm SCL from growth of captive *L. kempii*.

Von Bertalanffy parameter estimates for this study were near the mean maximum size observed on the nesting beach as calculated by Schmid and Witzell (1997), with an asymptotic length ( $a$ ) of 69.4 cm SCL and upper 95% confidence interval of 72.6 cm (Table 4). Using 62.5 cm SCL as the size-at-maturity estimate yields an age at sexual maturity of 9 years for *L. kempii* (Fig. 26). However, the von Bertalanffy growth equation indicates that *L. kempii* reach 20 cm SCL (juvenile size) after only one year. Based on skeletochronological data, it was assumed that *L. kempii* remain in the pelagic zone (5 - 20 cm SCL) for two years (Zug, 1990). The TEWG (1998) also estimated that *L. kempii* require two years to reach ca. 20 cm SCL based on a linear regression of hatchling production and juvenile strandings one, two, three and four years later. Consequently one year was added to the age-at-maturity estimate for an age-at-maturity of 10 years, with a resulting 95% confidence interval of 8 to 13 years.

### **Mortality and Life History**

Mortality as used in the population model is life-history stage specific. For this reason, the amount of time required to pass through each stage is critical to the model. Life history and number of years spent in each stage are described below.

Kemp's ridley life-history was divided into five stages based upon physiological and habitat utilization data. Ogren (1989) defined **four** life history stages, generally described in the literature as:

- 1) hatchling;
- 2) pelagic post-hatchling - the "lost-year" period of unknown length;
- 3) juvenile - near-shore, benthic individuals generally ranging from 20 to 60 cm SCL; and
- 4) adult - sexually mature individuals, generally greater than 60 cm SCL.

Developmental data collected for *L. kempii* suggest that Ogren's "juvenile"

life-history stage be further subdivided into prepubescent and pubescent stages based upon levels of circulating plasma testosterone (Coyne and Landry, 2000). For purposes of the model, these individuals will be referred to as juveniles and subadults, respectively.

Chaloupka and Zug (1997) suggested that *L. kempii* display polyphasic growth that may reflect growth spurts representing ontogenetic shifts in diet and habitat use. For example, Chaloupka and Zug suggest that the first shift, circa 20 cm SCL, coincides with a Kemp's ridley migration from a pelagic to coastal benthic habitat. A second change in growth occurs at approximately 40 cm SCL, providing additional support for distinguishing between juveniles and subadults.

A distinction between juveniles and subadults is also supported by Kemp's ridley stranding data. Figure 27 provides a cumulative frequency distribution of strandings along the U.S. Atlantic and Gulf of Mexico coasts. Portions of the distribution curve that fit a straight line indicate size ranges of turtles that strand at a similar rate, presumably subject to the same mortality pressures because of similar life-history strategies. Curves between these straight lines (grey areas) may represent transition periods between stages or a size range over which one stage begins and completes recruitment to the next stage. Five distinct ranges with similar stranding rates are apparent, presumably corresponding to the five life-history stages (hatchling, post-hatchling, juvenile, subadult and adult). Details of each stage are discussed below.

**Hatchling:** This stage includes the period from when eggs are first laid until hatchlings reach the water. Hatchling mortality is included in the model variable *Hatch Rate* (Table 3). Mortality of hatchlings after release from the beach is assumed a part of first-year pelagic post-hatchling mortality.

**Pelagic Post-hatchling:** Commonly referred to as the "lost-year", pelagic post-hatchlings are defined as those *L. kempii* found in the pelagic zone, presumably from the

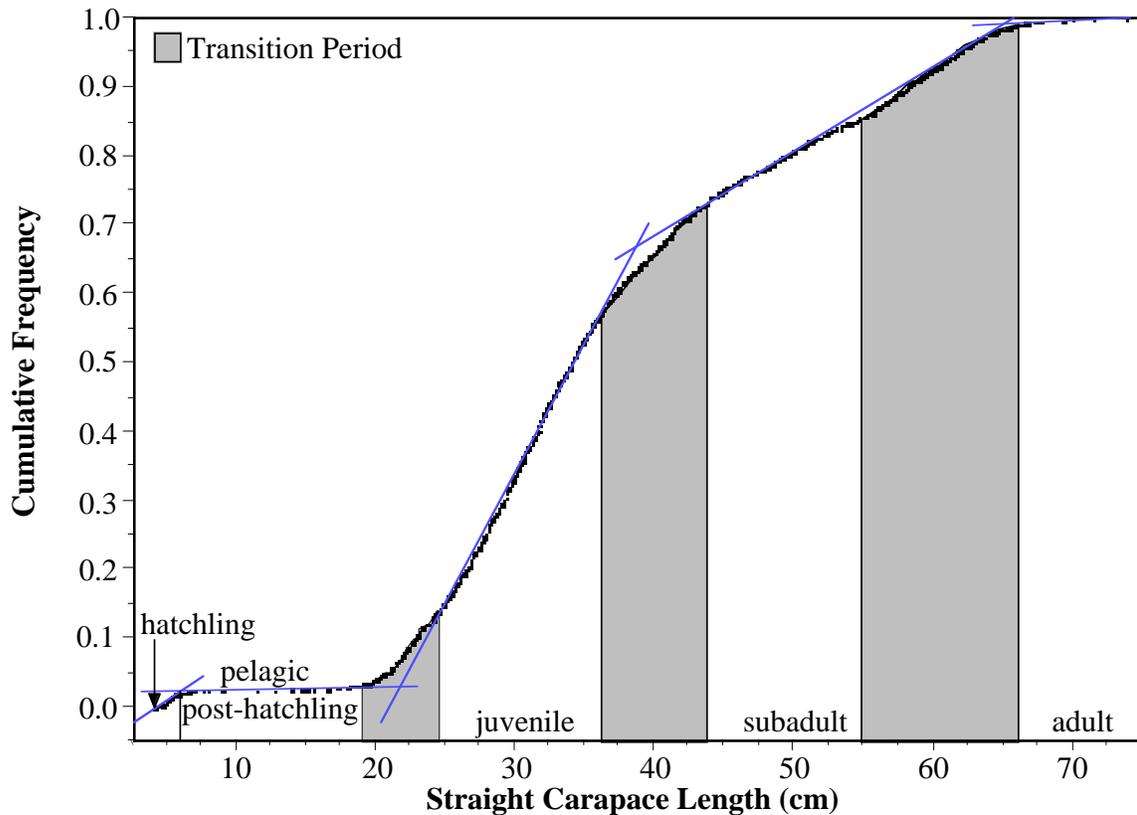


Figure 27. Cumulative frequency distribution of Kemp's ridley strandings by size (cm SCL) in the U.S. during 1986 - 1996 (NMFS Sea Turtle Stranding and Salvage Network). Straight lines represent individuals in the same life history stage subject to similar mortality pressures. Grey areas represent transitional size ranges at which one stage recruits to the next.

time they reach pelagic habitat until recruiting to near shore habitat circa 20 cm SCL (Ogren, 1989). More specifically, cumulative frequency distribution of Kemp's ridley strandings suggests that individuals of this stage begin recruiting to the nearshore environment at approximately 19 cm SCL and that recruitment is complete by 24 cm SCL (Fig. 27). Of 385 live *L. kempii* captures, the smallest individual encountered in near-shore environs during this study (1992 - 1997) was 19.6 cm SCL.

The von Bertalanffy growth equation suggests that *L. kempii* pass through the

pelagic post-hatchling stage in one year (Fig. 26). However, age estimates of juvenile *L. kempii* based on skeletochronology techniques indicate that individuals first appearing nearshore are two years old (Zug, 1990). The growth curve generated here does not include individuals less than 15 cm SCL and may not accurately reflect the growth of this stage; therefore, Zug's estimate of two years was used for the model.

**Juvenile:** This stage represents those near-shore individuals in a prepubescent state, exhibiting little or no sexual development as defined by low levels of circulating testosterone (Coyne and Landry, 2000). These individuals are approximately 20 - 40 cm SCL. Cumulative frequency distribution of Kemp's ridley strandings (Fig. 27) suggests that individuals recruit to this stage between 19 and 24 cm SCL. Recruitment to the subadult stage begins at approximately 37 cm SCL and is complete by 43 cm SCL. The von Bertalanffy growth interval equation suggests that *L. kempii* remain in the juvenile stage for approximately two years (Fig. 27). Observed growth of individuals captured during the course of this study supports this observation (Fig. 28). Individuals whose mean capture-recapture measurements ranged between 20 - 30 cm SCL (n = 27) and 30 - 40 cm SCL (n = 4) exhibited growth rates of 12.2 and 8.1 cm/yr, respectively, yielding 20 cm of growth in two years.

**Subadult:** This stage is equivalent to what many authors refer to as large or late juveniles (Chaloupka and Musick, 1997; Musick and Limpus, 1997). Subadult *L. kempii* were defined as those near-shore individuals in a pubescent state, undergoing sexual development (represented by increasing male circulating testosterone) in preparation for adulthood or approximately 40 cm SCL to adult. Cumulative frequency analysis of *L. kempii* strandings suggests that individuals begin and end recruitment to the subadult stage between 37 and 43 cm SCL (Fig. 27). Although this portion of the cumulative frequency curve is least distinctive, recruitment of subadults to the adult stage appears to begin at approximately 55 cm SCL and completes circa 65 cm SCL. The starting adult

recruitment size of 55 cm SCL seems reasonable as it is the minimum nesting size reported by Márquez (1994). The von Bertalanffy growth equation suggests that *L. kempii* spend approximately six years as subadults assuming a size-at-maturity of 62.5 cm SCL (Fig. 27). Limited capture-recapture size data (n = 3) indicate that 40 - 50 cm SCL individuals exhibit a growth rate of 6 cm/yr. Unfortunately, no recapture data were available for 50 - 60 cm SCL *L. kempii*.

**Adult:** Adults are those ridleys that have reached sexual maturity and are physiologically able to reproduce. Cumulative frequency analysis of *L. kempii* strandings suggests that individuals begin recruitment to the adult stage at approximately 55 cm SCL (size of the smallest reported nesting turtle) and complete recruitment by 65 cm SCL (Fig. 27). As discussed previously, the exact size at which Kemp’s ridley sea turtles reach adulthood is uncertain, but mean size-at-maturity was assumed to be 62.5

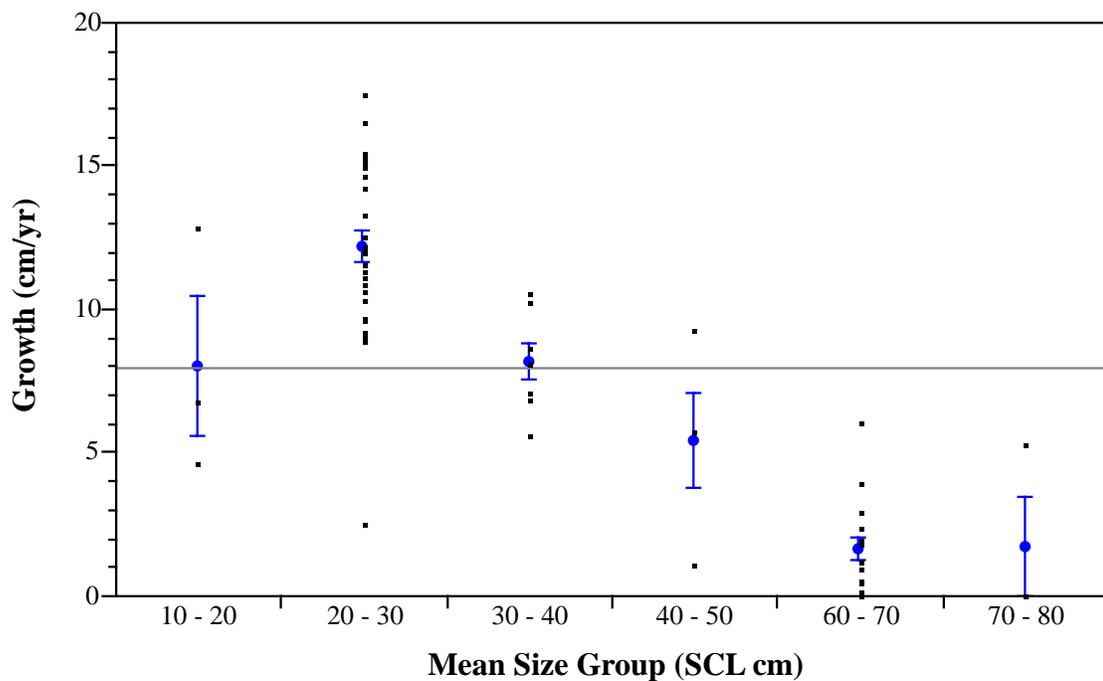


Figure 28. Mean growth (cm/yr) and standard error of recaptured Kemp’s ridleys within mean capture-recapture size classes (SCL cm;  $r^2 = 0.758$  ).

cm SCL and age-at-maturity approximately 10 years. The adult population was modeled without age structure, with individuals going into an “adult pool”.

**Mortality:** Márquez (1994) suggested survival rates of 0.589 for post-hatchlings, 0.565 for juveniles, 0.445 for subadults (pre-adults) and 0.421 for adults (annual mortality 0.411, 0.435, 0.555 and 0.579, respectively). These estimates were based upon a least-squares regression of hatchlings produced compared to number of nests observed 7 years later (presumed age-at-maturity for the studies; Table 5). Heppel *et al.* (1996b), in a model to evaluate headstarting as a management tool, estimated *L. kempii* survival using survival estimates derived for similar sized loggerheads (Frazer, 1986). Estimates were based upon 8, 12 and 16 years to maturity. Equivalent annual mortality rates for 12 years to maturity are given in Table 5 and assume that hatchlings, juveniles and

Table 5. Annual mortality rate estimates for Kemp’s ridley sea turtles used in this and other studies. Changes in survival rates over time are due to management efforts: pre-1978 = no management; 1978 - 1990 = beach protection; post-1990 = beach protection and Turtle Excluder Device regulations.

	Pelagic Post-hatchling	Benthic		Adult	Source
		Juvenile	Subadult		
	0.4110	0.4350	0.5550	0.5790	Marquez, 1994; 7 years to maturity
pre-1978	0.8375	0.3796	0.3242	0.2575	Heppel <i>et al.</i> , 1996b; based on loggerhead study; 12 years to maturity
1978-1990	0.6750	0.3796	0.3242	0.2575	
post-1990	0.6750	0.3796	0.2270	0.1802	
pre-1990	0.5580	0.3430		0.1590	TEWG, 1998; juveniles and subadults treated as one stage; 10 years
post-1990	0.5580	0.2570		0.1100	
pre-1990	0.5580	0.4770	0.3070	0.1590	this study; 10 years to maturity
post-1990	0.5580	0.3220	0.2260	0.1100	

subadults spend one, four and seven years in each stage, respectively.

The TEWG (1998) calculated proportionate survival to be 0.442 for hatchlings (age 0-1 year), 0.657 for benthic immatures (age 2-9 years) and 0.841 for adults (age 10+ years) up to 1990 (pre-TED regulations) and 0.442, 0.753 and 0.89, respectively, after 1990 (TED regulations enacted). Equivalent mortality rates are given in Table 5. These estimates were derived from a least-squares fit criteria of predicted versus observed nests (1978 - 1995), after first estimating instantaneous mortality for benthic immature *L. kempii* (juveniles and subadults in this model) derived from stranding rates in the NMFS Sea Turtle Stranding and Salvage Network.

Survival/mortality estimates calculated by the TEWG (1998) were used in this model and likely represent the best available estimates for *L. kempii*, given current understanding and data availability. However, any change in behavior or habitat use between juvenile and subadult *L. kempii*, as previously discussed, would subject the two life-history stages to different mortality pressure. Hence, mortality rates were recalculated for juvenile and subadult *L. kempii* that are mathematically equivalent to those calculated for benthic immatures by the TEWG (1998). New pre- and post-1990 mortality estimates for juvenile and subadult *L. kempii* were derived from a least-squares fit of predicted versus observed nests (1978 - 1998). Post-pelagic and adult mortality rates were held constant while restricting the cumulative mortality product of two juvenile and six subadult years equal to the product of eight years of TEWG benthic immature mortality (i.e.,  $juvenile\ mortality^2 \cdot subadult\ mortality^6 = benthic\ immature\ mortality^8$ ).

## **Model Results and Evaluation**

A functional version of the conceptual Kemp's ridley population model (Fig. 22) was created using Stella Research Software version 5.1.1 for Power Macintosh (Appendix B). Estimated variables used in the model were as discussed in the previous section (see Model Formulation and Quantification). A 50 year time-span, commencing in 1978, was used for all model runs with a one-year time step. This allowed limited validation of the model's ability to predict actual number of nests reported through 1999.

### **Seeding the Model**

Initial number of individuals in each age-class in 1978 were estimated by back-calculating from the number of nests reported in subsequent years, based upon clutch frequency, remigration rate and mortality. For example, to estimate the number of adults in 1978, number of nests reported that year (924) was divided by clutch frequency (3) and remigration rate (0.768). This resulted in an estimate of 401 adult females in the population in 1978. An equal number of adult males was assumed based on a 1F:1M sex ratio.

To estimate the number of age-class ten (sixth year subadult) individuals in 1978, the number of adult females in 1979 (414) was estimated as outlined above. The number of 1979 neophyte nesters was then estimated by subtracting the number of adult females in 1978, taking into account one year of adult mortality, from the number of adult females in 1979  $[414 - (401 \cdot (1 - 0.1590)) = 77]$ . Number of age-class ten females in 1978 was then calculated  $(77 \div (1 - 0.3070)) = 111$  as the number of 1979 neophyte nesters after back-calculating one year of subadult mortality. An equal number of same-age males was again assumed. This method of seeding the model is auto-correlating, resulting in predicted nests exactly equaling actual nests reported for a number of years equal to the number of age-classes seeded. In this case 10 years-to-maturity was

assumed, meaning that for the first 10 years of the model predicted nests were expected to exactly equal actual nests. Thus, model validation statistics were only performed on number of nests from 1989 - 1999.

### Baseline Model

The baseline model was run with all variables set as described previously and a sex ratio of 1F:1M. Results suggest an exponential increase in the number of nests (Figs. 29 and 30), as has been observed in other modeling efforts (TEWG, 1998). Observed versus expected nests from 1989 through 1999 were strongly correlated

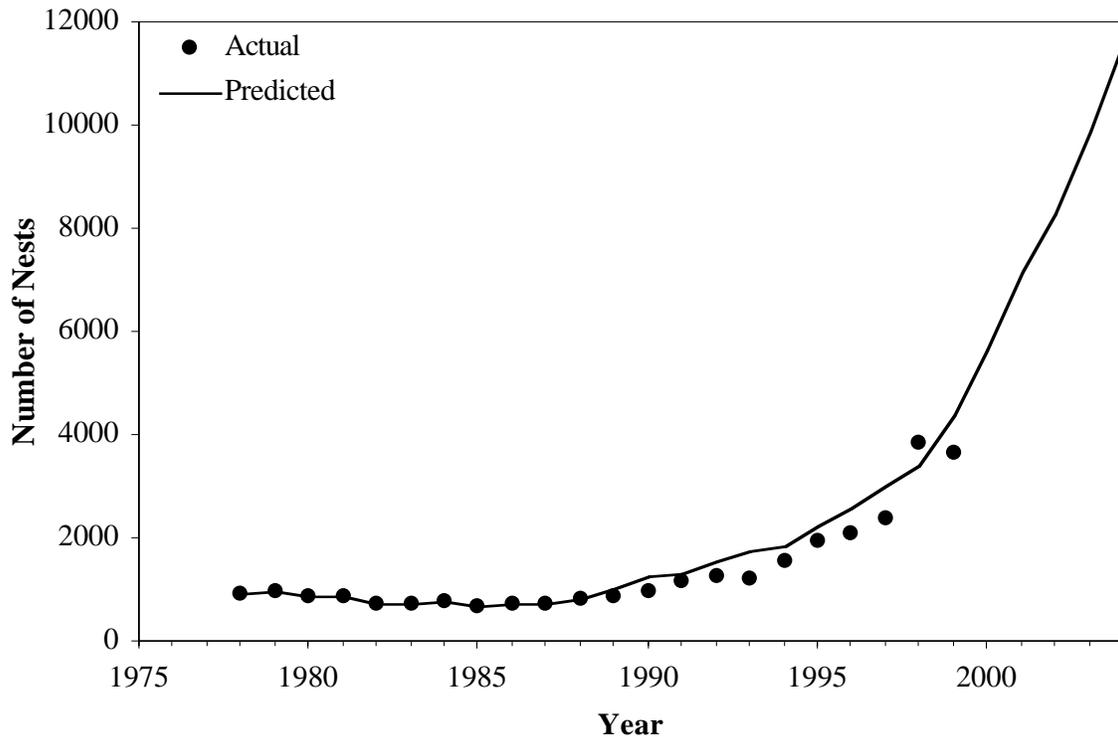


Figure 29. Actual number of Kemp's ridley nests reported each year at Rancho Nuevo, Mexico versus that predicted by the population model. Correlation coefficient for 1989 - 1999 = 0.955.

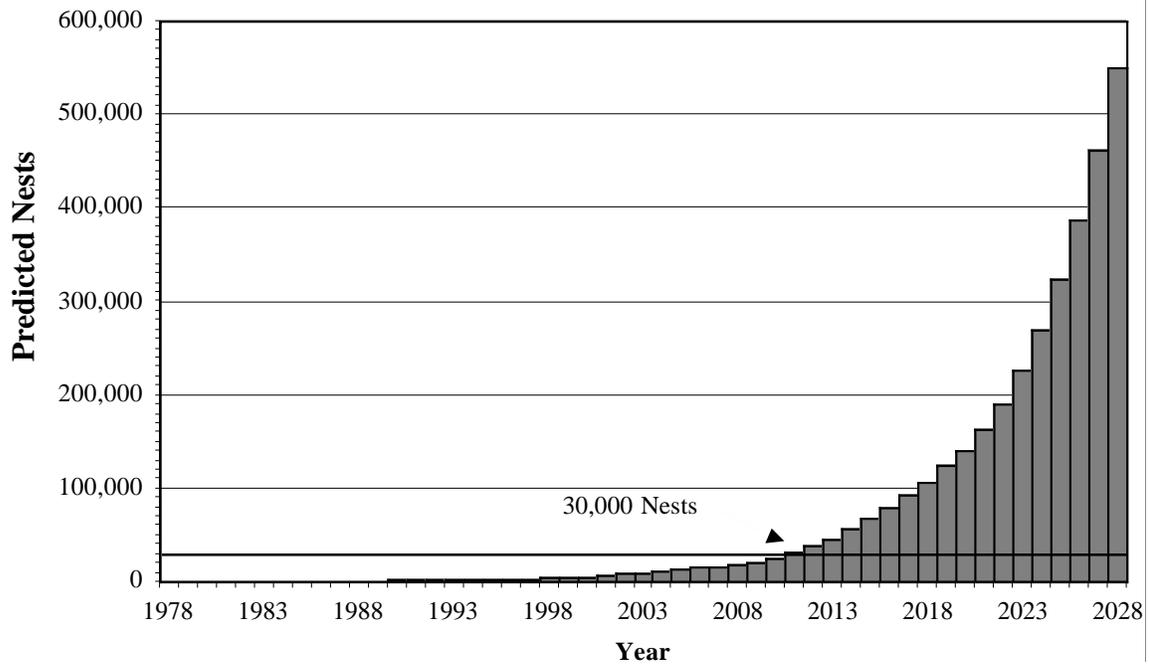


Figure 30. Number of Kemp's ridley nests predicted by the baseline model projected from 1978 through 2028.

(correlation coefficient = 0.955). Despite underestimating a relatively large increase in number of nests in 1998, the model appears to perform well over this span. Predicted number of nests increased to 549,934 after a 50-year run of the model, through the year 2028 (Fig. 30).

For reference, one criterion set forth by The Kemp's Ridley Recovery Team (USFWS/NMFS, 1992) for downlisting this species to Threatened status under the Endangered Species Act is to attain a population of at least 10,000 females nesting in a season. In terms of this model, 10,000 nesting females is approximately equivalent to 30,000 nests laid (number of nesting females • 3 clutches per female). Given certain model caveats and uncertainty surrounding many model variables, results suggest that this goal can be achieved by the year 2011.

## Sensitivity Analysis

Sensitivity analyses were performed to identify those variables most likely to cause error in the model if estimated incorrectly. These analyses can also be used to identify the most critical life-history parameters in terms of population management. For example, one can assess the importance of accurately estimating a particular variable or what impact a real change in a given parameter may have on the population.

Sensitivity analysis was accomplished by individually altering each static model variable by +/-10 and 20% from the baseline value and comparing the resulting model output (Fig. 31). Analyzed model variables included remigration rate, clutch frequency,

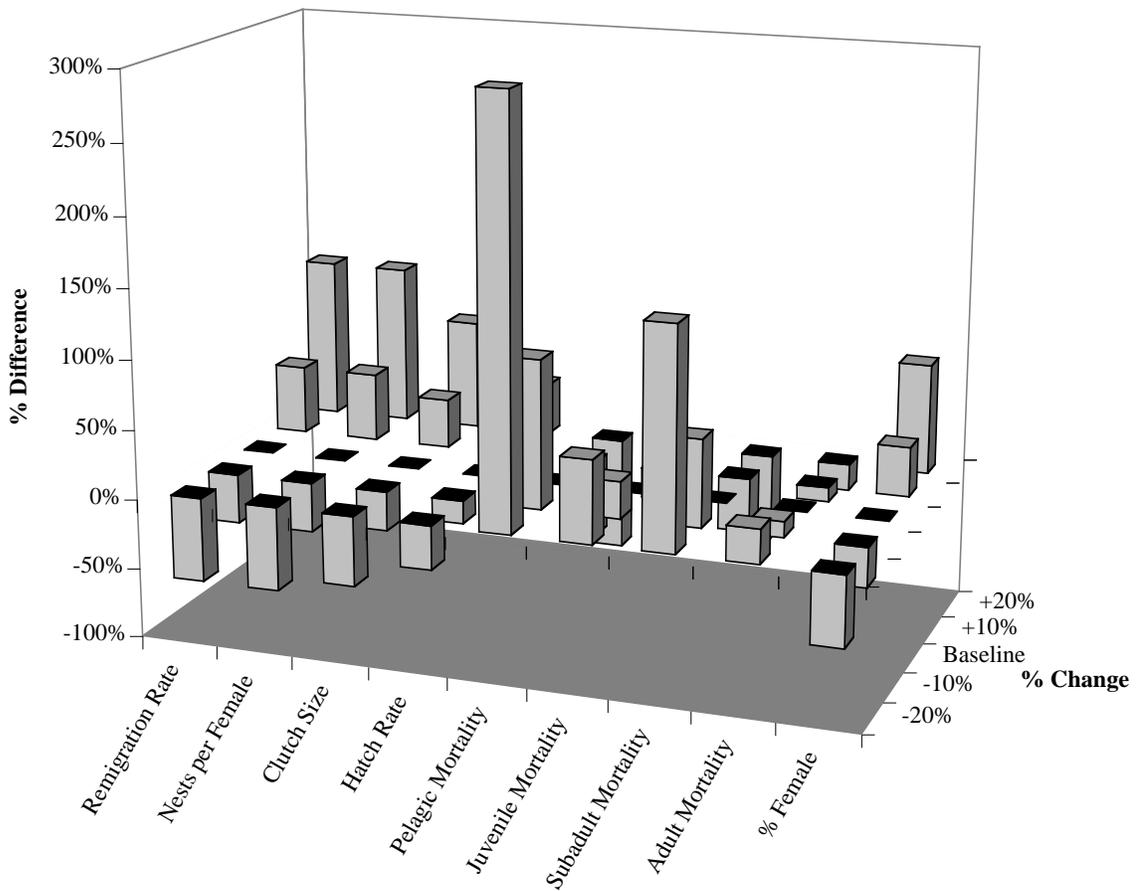


Figure 31. Percent difference from baseline number of nests as a result of altering specified model variables by +/-10 and 20%.

clutch size, hatch rate, pelagic mortality, juvenile mortality, subadult mortality, adult mortality and sex ratio. Table 6 identifies those variables yielding the greatest percent difference from baseline in estimated number of nests after 50 years.

The most sensitive model variable appeared to be pelagic post-hatchling mortality. A 20% decrease in the variable resulted in more than a 300% increase in number of nests from baseline after 50 years. Other mortality changes that resulted in relatively large differences in the model, defined as a 50% deviation in number of nests from baseline after 50 years, were -20% subadult mortality (+156% nest count), -10% pelagic post-hatchling mortality (+106% nest count), +20% pelagic post-hatchling mortality (-80% nest count), -10% subadult mortality (+62% nest count), +20% subadult mortality (-62% nest count), -20% juvenile mortality (+59% nest count) and +10% pelagic post-hatchling mortality (-55% nest count).

Other model variables exhibiting sensitivity to change were +20% remigration rate (+115% nest count), +20% clutch frequency (+115% nest count), +20% clutch size (+79% nest count), +20% percent female (+79% nest count), -20% remigration rate (-60% nest count), -20% clutch frequency (-60% nest count), -20% clutch size (-50% nest count) and -20% percent female (-50% nest count).

Number of nests relative to baseline increased and decreased in relation to the proportion of female hatchlings produced. Only the 20% shifts in sex ratio resulted in relatively large changes in predicted number of nests after 50 years. A 79% increase in number of nests over baseline was observed when percent female was changed by +20% (Fig. 31; Table 6). Conversely, predicted number of nests decreased by 50% as a result of a -20% change in percent female. A 10% change, up and down, in percent female resulted in a 35% increase and 28% decrease in number of nests, respectively. Potential changes in fecundity related to adult sex ratio were not included in the sensitivity analyses.

Table 6. Percent change (%Diff) in number of nests from baseline as a result of altering model variables by +/-10 and 20% (Change). Variables are sorted in order of greatest to lowest absolute percent change (Abs %Diff) in number of nests.

<b>Variable</b>	<b>Change</b>	<b>Value</b>	<b>%Diff</b>	<b>Abs</b>	<b>%Diff</b>
Pelagic Mortality	-20%	0.456	301%	301%	
Subadult Mortality	-20%	0.1808	156%	156%	
Remigration Rate	+20%	0.922	115%	115%	
Clutch Frequency	+20%	3.6	115%	115%	
Pelagic Mortality	-10%	0.507	106%	106%	
Pelagic Mortality	+20%	0.66	-81%	81%	
Clutch Size	+20%	114	79%	79%	
Percent Female	+20%	0.6	79%	79%	
Subadult Mortality	-10%	0.2034	62%	62%	
Subadult Mortality	+20%	0.2712	-62%	62%	
Remigration Rate	-20%	0.614	-60%	60%	
Clutch Frequency	-20%	2.4	-60%	60%	
Juvenile Mortality	-20%	0.2576	59%	59%	
Pelagic Mortality	+10%	0.609	-55%	55%	
Clutch Size	-20%	76	-50%	50%	
Percent Female	-20%	0.4	-50%	50%	
Remigration Rate	+10%	0.845	49%	49%	
Clutch Frequency	+10%	3.3	49%	49%	
Juvenile Mortality	+20%	0.3864	-39%	39%	
Subadult Mortality	+10%	0.2486	-39%	39%	
Hatch Rate	+20%	0.846	37%	37%	
Remigration Rate	-10%	0.691	-35%	35%	
Clutch Size	+10%	104.5	35%	35%	
Percent Female	+10%	0.55	35%	35%	
Clutch Frequency	-10%	2.7	-35%	35%	
Hatch Rate	-20%	0.564	-31%	31%	
Clutch Size	-10%	85.5	-28%	28%	
Percent Female	-10%	0.45	-28%	28%	
Juvenile Mortality	-10%	0.2898	26%	26%	
Adult Mortality	-20%	0.088	24%	24%	
Juvenile Mortality	+10%	0.3542	-22%	22%	
Adult Mortality	+20%	0.132	-19%	19%	
Hatch Rate	+10%	0.7755	18%	18%	
Hatch Rate	-10%	0.6345	-16%	16%	
Adult Mortality	-10%	0.099	11%	11%	
Adult Mortality	+10%	0.121	-10%	10%	

### Sex Ratio Effects

The effect of sex ratio on the model was assessed across sex ratios ranging from 3F:1M to 1F:3M (Table 7). The number of individuals initially placed in each age-class bin was equally distributed between males and females. Sex ratio changes were applied to hatchlings dividing them into male and female pelagic post-hatchlings (Fig. 22). This is analogous to a management program controlling annual hatchling sex production beginning in 1978. In all cases, the adult pool of animals started as 1F:1M to reduce model complexity, but the adult population achieved applied test sex ratios within 14 to 16 years.

Sex ratios were tested under two scenarios: one assuming no fecundity effects related to the relative number of adult male and female *L. kempii*, and the other assuming fecundity effects as discussed in the previous section. Under a scenario of no fecundity effects, the number of nests increased by 144 and 271% over baseline after 50 years, respectively, for sex ratios of 2F:1M and 3F:1M (Fig. 32). Male biased sex ratios yielded a 72% decrease in number of nests for 1F:2M and an 88% decrease for 1F:3M.

Table 7. Sex ratios tested in the Kemp's ridley population model under scenarios with and without male-mediated fecundity. Number of nests is percent difference from baseline after 50 year model run.

Sex Ratio	% Female	Number of Nests	
		No Male-mediated Fecundity	Male-mediated Fecundity
3F:1M	75	271%	-20%
2F:1M	66	144%	-5%
1F:1M	50	-	-
1F:2M	33	-72%	-36%
1F:3M	25	-88%	-62%

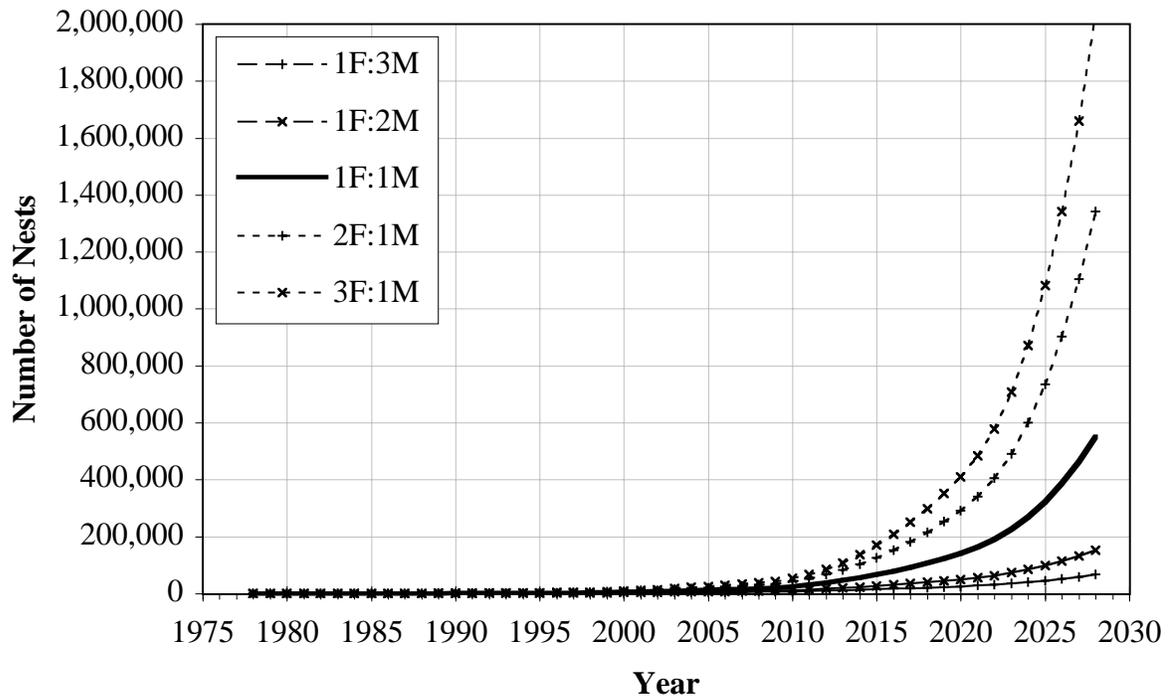


Figure 32. Number of nests resulting from five sex ratios applied to the population model, given a 50 year run and no sex ratio fecundity effects.

Under fecundity effect scenarios, in which clutch size and frequency are proportional to percent adult female (Fig. 23), all four test sex ratio cases yielded fewer predicted nests after a 50 year model run than did the baseline model (Fig. 33). A female bias resulted in less reduction in predicted nests than did a male bias (Table 7). A sex ratio of 1.28F:1M (56.2% adult females) provided the greatest increase in number of nests (+3%) over the baseline sex ratio of 1F:1M (Fig. 34).

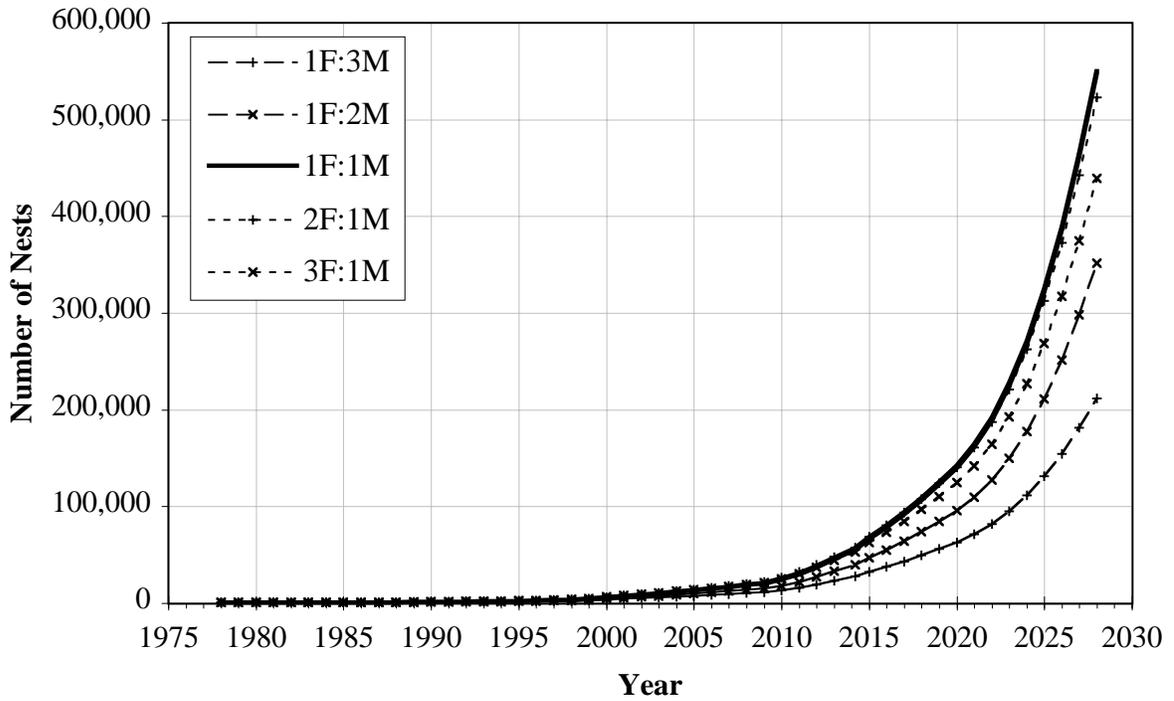


Figure 33. Number of predicted nests resulting from five sex ratios applied to the Kemp’s ridley population model after a 50 year run under a sex ratio fecundity effects scenario.

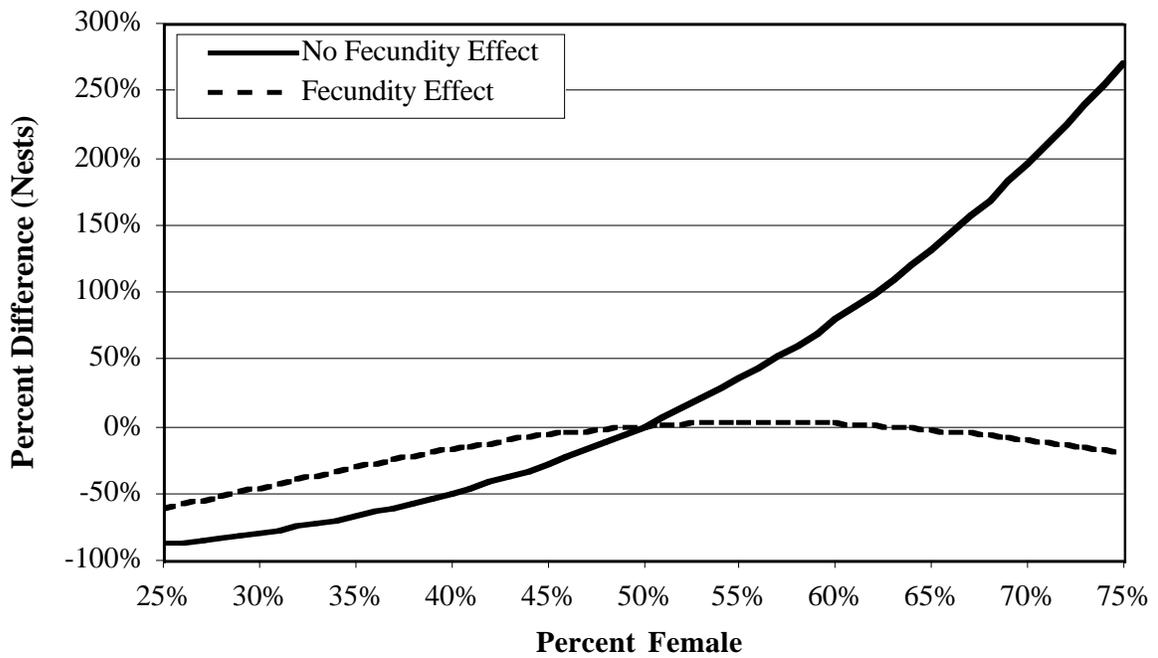


Figure 34. Percent change from baseline in predicted number of nests after 50 year model run with and without fecundity effects, across a range of sex ratios (1F:3M to 3F:1M).

## Discussion

The size and growth of any population depend on the annual number of births and deaths, and the timing of maturation, reproduction and death in each individual's life. This model was designed to be a generalized representation of the Kemp's ridley population, whose intent was to respond in a manner similar to that of the natural population. Attempting to predict exact population responses would be a difficult, if not impossible, task given that many aspects of *L. kempii* life-history remain unknown. Consequently, there are many assumptions related to development of this model that should be stated before discussion of the results.

### Caveats and Assumptions

*Deterministic versus stochastic modeling.* Many sea turtle studies have reported significant individual variability, particularly in growth rates, suggesting that including variable development rates in population models might be worthwhile (Bjorndal and Bolten, 1988; Ehrhardt and Witham, 1992; Klinger and Musick, 1995; Chaloupka and Limpus, 1996). However, life-history variation in this model was sacrificed for the sake of simplicity and insufficient knowledge. Some annual variation was introduced into the model using reported hatch rates from 1978 - 1998, but for simplicity hatch rate beyond 1998 was set as a constant value. Annual mortality rates were also considered to be constant, except for a one-time adjustment in 1990 to approximate a change in mortality as the result of TED regulations. In general, dynamic variables within the model were approximated as mean values. Therefore, model results should be considered a mean population response and do not include normally occurring annual and seasonal fluctuations.

*The model population is not density-dependent.* Regardless how large the model population becomes, it will continue to increase. In reality, it is likely that if the

population continues to increase one or more factors, or limiting resources, will act to control population growth. These factors can take many forms including competition for limited food resources or nesting beach space. For example, a Kemp's ridley model developed by the TEWG (1998) decreased hatching success as the population increased, under the assumption that there is an upper limit to the number of nests that can be protected each nesting season. They estimated that a maximum of 3,000 nests could be protected in corrals, with an additional 2,000 nests protected *in situ* by screening. Nests deposited beyond these 5,000 would be left undisturbed, which becomes a limiting factor because they would be subject to predation or other sources of mortality, resulting in a lower hatch rate. In general, density-dependence is poorly understood in sea turtles and it is likely to be some time, considering estimated historical population levels, before *L. kempii* become resource limited by anything aside from nesting beach space.

Therefore, density-dependent factors were not considered.

*Reported nest counts from the nesting beach do not represent actual total number of L. kempii nests.* Limited resources prevent the Kemp's ridley project from completely covering the full spatial and temporal nesting range each season. For this reason, an unknown number of nests is not counted each season. Number of nests reported was considered the best estimate available and assumed to represent a relatively consistent proportion of actual number of nests each year.

*Expanded coverage of nesting beaches.* The scope of the Kemp's Ridley Project has expanded over the years beyond the initial camp at Rancho Nuevo. Additional camps established in 1988, 1989 and 1996 increased both the effort expended and area covered, providing project personnel an opportunity to document nests in new locations. This may account for some, but certainly not all, of the increase in *L. kempii* nest counts in recent years (Fig. 29). Unfortunately, it is not clear how the actual number of nests counted each year has changed relative to increased effort. The model uses total number

of nests reported each year by the Kemp's Ridley Project from Tamaulipas, Mexico. *Number of nests may not be an accurate indicator of population status or size.* Annual nest counts represent one of the better understood and documented aspects of *L. kempii* life-history, but are they representative of the population as a whole? In theory, number of nests laid in a given year will be a function of the proportion of new recruits and remigrants of various year classes in the population (Limpus, 1995). However, it has been reported that individual sea turtle maturation times vary greatly, meaning that all cohorts hatching in a given year do not necessarily reach maturity at the same time. For example, Limpus *et al.* (1994a, 1994b) reported that eastern Australian green and loggerhead turtles do not begin to breed at a uniform size and display sex-specific differential growth rates during portions of development (Chaloupka and Limpus, 1997; Limpus and Chaloupka, 1997). It is unclear whether these differential growth rates translate into differing maturation times. Similar sex specific growth rates have not been positively identified for *L. kempii*; however, limited skeletocronology suggests that Atlantic and Gulf of Mexico cohorts exhibit differential growth, likely related to regional water temperature differences (Caillouet *et al.*, 1995; Zug *et al.*, 1997). Highly variable maturation times could introduce errors into population estimates based upon nest counts or other nesting beach indicators because some unknown proportion of the expected nesting population is not nesting. Reported nest counts were used in this model and assumed to be the best indicator of population size at this time.

*All adult females are equally fecund.* The model assumes that all adult females, regardless of age, exhibit the same fecundity. In other words, individual adult females have the same level of reproductive output throughout adulthood. In reality, factors related to adult productivity (clutch size, clutch frequency, remigration rate, etc.) are probably not consistent. There is evidence to suggest that neophyte and older adult female nesters are less productive than adult cohorts in their reproductive prime.

Limpus (1995) reported that female eastern Australian loggerhead sea turtles exhibit an increase in egg production with age as they mature from being new recruits to remigrants in their second and third breeding seasons. Most of these increases in egg production are the result of shortening the interval between breeding seasons and increasing the number of clutches laid per season. Increases in the number of eggs per clutch made little contribution (Limpus, 1995).

*Duration of the pelagic post-hatchling stage is unknown.* As discussed previously, a year was added to the one year estimate of the pelagic post-hatchling life-history stage (circa 20 cm SCL or less) derived from the von Bertalanffy growth equation developed for the model (Fig. 26). It has been generally accepted that *L. kempii* enter the neritic (near-shore) habitat after 1-2 years in the pelagic stage (Ogren, 1989; Zug, 1990). In addition, skeletochronology data suggest that *L. kempii* are approximately two years old at circa 20 cm SCL (Zug *et al.*, 1997).

Uncertainty regarding duration of the pelagic post-hatchling stage led to initiation of a program in 1996 by the NMFS Galveston Laboratory in coordination with the Instituto Nacional de la Pesca and the Gladys Porter Zoo. The goal of the program was to tag 10,000 hatchlings from each of two consecutive year-classes with internal wire tags (Caillouet *et al.*, 1997; Burchfield *et al.*, 1998). Logistical problems restricted tagging to only 3,346 individual in 1996. The program successfully wire tagged 10,002 hatchlings in 1997. Logistical problems prevented tagging in 1998 and another 10,010 hatchlings were wire tagged in 1999 (Cannon and Higgins, *in press*; Jaime Peña, pers. comm.). It is hoped that these efforts will help quantify the pelagic post-hatchling stage, after which these individuals begin to recruit to and are identified in near-shore habitats.

*Equal treatment of Atlantic and Gulf of Mexico cohorts.* It has been suggested that *L. kempii* in coastal U.S. Atlantic waters, where water temperatures are cooler and food resources less abundant, grow more slowly than their Gulf of Mexico counterparts

(Caillouet *et al.*, 1995). However, these potential differences have not been quantified and it is unclear when or how often *L. kempii* found in Atlantic waters return to the Gulf of Mexico. For purposes of the model, all individuals in the population were treated equally with regard to growth.

*Remigration Rate.* Actual remigration intervals or periodicity may be larger than are currently understood. Errors in calculating this variable may result from tag loss, annual variation or incomplete spatial and temporal nesting beach survey coverage (Limpus, 1992; Miller, 1997).

*No life expectancy limit.* Maximum life expectancy was not placed on individual turtles moving through the model since this variable has not yet been quantified for *L. kempii*. For purposes of the model, individuals enter an adult pool. Turtles are removed from the pool at a set mortality rate, with no upper limit on time spent in the adult pool.

*Male mediated fecundity.* Although there is anecdotal evidence that availability of adult males plays a role in sea turtle reproduction beyond simple copulation and fertilization, there is no quantifiable support for the notion that relative abundance of adult males and females impacts reproductive output or success (Wood and Wood, 1980; Mrosovsky and Godfrey, 1995). For this reason, estimates of male-related fecundity in the model represent a “best guess”. The method of deriving this relationship was chosen for simplicity and ease of understanding, rather than a result of quantitative analysis (Fig. 23).

### **Model Performance**

Given the caveats and uncertainty surrounding many variables, the model appears to perform well over the time span for which actual nest counts are available (1989 - 1998). Estimated nests were strongly correlated with actual nest counts during this time. Recorded nesting counts from 1978 - 1988 were excluded from comparison

with model results because the method used to seed the model resulted in predicted nests that equaled actual nests reported for the first 10 years of the model (estimated time to maturity).

Results of the model suggest that the benchmark set for downlisting *L. kempii* from Endangered to Threatened status (10,000 nesting females in a season) can be achieved by the year 2011 (Fig. 30). A population model developed by the TEWG (1998) suggested a more conservative recovery goal of 10,000 nesting females by the year 2020. Differences in these model predictions are partially the result of the current model not taking density-dependent factors into account. As mentioned previously, the TEWG model placed an upper limit of 5,000 nests that can be protected, rendering all others laid in a single season subject to natural predation.

### **Sensitivity Analysis**

Sensitivity of the model to variations in parameter values was tested by comparing model predictions to what is known about sea turtle population dynamics. This exercise also allows one to explore potential effects of different management scenarios on the long-term survival of *L. kempii*. For example, Crouse *et al.*, (1987) used sensitivity analyses of a loggerhead model to suggest that current management practices appear to focus on the least responsive life stage, eggs on the nesting beach, when protection can be improved if more responsive stages are targeted. It is possible to obtain conflicting recommendations from similar models, particularly if they are based upon different populations of the same species. Heppel *et al.* (1996a) suggested that reducing mortality in *C. caretta* greater than 70 cm CCL was essential to stock viability. Conversely, Somers (1994) concluded that protection of eggs and hatchlings was most important, possibly because the population in question had higher egg/hatchling mortality (Chaloupka and Musick, 1997).

Many variables exhibiting the greatest impact on the current model's results were related to changes in mortality (Table 6). For example, a 20% reduction in pelagic post-hatchling mortality rate resulted in a 301% increase in the number of predicted nests after 50 years. A 10% reduction in pelagic post-hatchling mortality yielded a 106% increase in predicted nests, while mortality increases of 10 and 20% reduced the population by 55 and 81%, respectively. These results suggest that protection of *L. kempii* pelagic post-hatchlings could yield the greatest population increase.

Unfortunately, because so little is known about pelagic post-hatchlings, this is probably not a viable option. A more likely scenario, reducing subadult mortality, also had a large impact on model results, with a 20% decrease yielding a 156% increase in estimated nests after 50 years. A 20% decrease in juvenile mortality resulted in a 59% increase in estimated nests. Reducing mortality of reproductively mature *L. kempii* had little impact on the model, perhaps because a 10 or 20% change yields little difference in the already low adult mortality rate (0.110; Table 6).

In general, variables that have a large impact on the model suggest: 1) potential areas for concern (e.g., important life-stages to protect); 2) good candidates for focused management effort; or 3) simply that accurate assessment of the variable in question is critical because it appears to have such a large effect on model results. These model results should be treated with extreme caution because estimated mortality rates represent one of the least reliable model variables and appear sensitive to change.

Analysis of other, non-mortality related model variables suggests that only the greatest changes (+/- 20%) appreciably (>50%) alter estimated number of nests after 50 years. Changes to remigration rate and clutch frequency exhibited the greatest deviation in model results, each yielding a 115% increase, approximately three times less than the greatest increase observed by reducing mortality. While managers are not likely to influence remigration rate and clutch frequency, it does suggest that accuracy of these

variables is critical to the model.

Changes in adult mortality and hatch rate yielded the smallest difference in model results. This begs the questions, how important are current management practices to recovery of Kemp's ridley sea turtles? Could recovery efforts be better focused? Current practices on the nesting beach primarily impact the variable hatch rate in this model. It is not surprising that increasing hatch rate yields only a small benefit to the population because the variable is already relatively high. However, a 20% decrease in hatch rate (from 0.705 to 0.564) yields only a 31% decrease in the model population after 50 years. This suggests that current management recovery efforts, while certainly effective to some extent, would be better focused on other life-history stages or management strategies.

A recently proposed management strategy is creation of a protected area off the Texas coast to safeguard adult *L. kempii* that have come there to nest or are in route to their Mexican nesting beach. Would money, effort, and most important, publicity, not be better focused on what appear to be life-history stages that would yield a greater return on investment? Do these efforts serve to distract from better alternatives? Based upon the current model an obvious choice appears to be added protection for pelagic post-hatchling ridleys. Unfortunately this is a stage we know little to nothing about and probably have little ability to protect.

Based upon model results other factors can also be considered. For example, protecting adult *L. kempii* off the Texas coast would play an important role in fostering establishment of an alternative nesting beach, in addition to increasing the population as a whole. This area is also host to a seasonally high concentration of adult *L. kempii* as they migrate to their nesting beaches, making them more susceptible to anthropogenic mortality (such as shrimp trawling in the area) during that time.

## Sex Ratio Dynamics

Studies reporting *L. kempii* sex ratios other than 1F:1M have frequently detected a female bias, ranging from 1.4F:1M to 3.2F:1M (Table 2). On the surface, a naturally occurring female bias seems an obvious strategy that a population can utilize to increase productivity. From a management perspective, it presents a potential and very attractive tool to aid in the recovery of threatened or endangered species. However, as noted by other authors, sea turtle sex ratio dynamics may not be that simple, and their manipulation may lead to undesirable consequences (Mrosovsky and Godfrey, 1995; Lovich, 1996).

The assertion that adult sex ratios are naturally biased in some turtle species, as has been shown in numerous studies, has profound implications for any program attempting to manipulate sex ratio to manage a population (Morreale *et al.*, 1982). A major concern would be the impact of such a program on the reproductive ecology of a species, specifically as it relates to effects of multiple paternity, sperm competition, fertility, and intra-specific competition on population persistence.

If one assumes that proportion of available males has little or no impact on production, as long as enough males are available for mating, then the current model confirms an increased proportion of females in the population will dramatically enhance hatchling production (Fig. 34). Under this scenario, nest production increases as a power of percent female hatchlings produced, yielding a 271% increase in predicted nests after 50 years with a sex ratio of 3F:1M and an 81% decrease with 1F:3M:

$$\% \text{ Nest Increase} = 0.3005 \cdot (\% \text{female hatchlings produced})^{2.2615}$$

However, an increasing body of knowledge suggests that there is a benefit to having some minimum proportion of adult males in a sea turtle population. For example, multiple copulations or more time spent copulating can potentially yield a greater number of fertilized eggs, thereby increasing fecundity (Wood and Wood, 1980;

Madsen *et al.*, 1992; Mrosovsky and Godfrey, 1995). In addition, copulation with multiple males may increase clutch viability, improving overall genetic health of a clutch or clutches (Harry and Briscoe, 1988; Sugg and Chesser, 1994; Kichler *et al.*, 1999).

It is well established that several turtle species exhibit multiple paternity (Harry and Briscoe, 1988; Kaufmann, 1992; Galbraith *et al.*, 1993). Coupled with the sperm storage ability possessed by many turtles (Gist and Jones, 1989), multiple paternity and sperm competition must be considered significant adaptations in their reproductive strategy. This suggests that multiple paternity and sperm competition may be very important in the persistence of populations and cautions against manipulating the sex ratio of turtle populations to produce an excess of females (Lovich, 1996).

Under this scenario, reproductive output and success are partially a function of male availability. Assuming a linear relationship between percent increase in adult males and reproductive output (Fig. 23), the model suggests that strong female bias actually results in a decrease in the population. The hatchling sex ratio producing the greatest rate of reproductive return was 56.2% female (1.28F:1M; Fig. 34). A strong female bias (3F:1M) results in a 20% decrease in the population while a male bias (1F:3M) yields a 62% decrease. It should be understood that the actual relationship between male availability and female reproductive output is not known! These results further stress that extreme care should be taken when considering the manipulation of sex ratio as a sea turtle conservation strategy.

## CHAPTER IV

### SUMMARY AND CONCLUSIONS

#### Summary

This study establishes a testosterone radioimmunoassay (RIA) sexing criteria for Kemp's ridley sea turtles (*Lepidochelys kempii*). The results of RIA analysis in conjunction with limited validation utilizing laparoscopic examination suggest that at-sea captured, non-nesting female *L. kempii* yield plasma testosterone (T) concentrations of 12 pg/ml or less, while male counterparts yield 18 pg/ml or greater. Sex was not validated for 10 of 247 blood samples falling between 12 and 18 pg/ml plasma T and, as such, gender of these turtles was considered indeterminate.

The RIA sexing criteria should be treated with caution, particularly when used with other studies, as assay results may vary between laboratories (Gregory and Schmid, 1998; Wibbels *et al.*, in press). Capture stress and regional or seasonal changes in plasma T may also result in sexing errors. For example, a male juvenile captured during the winter could exhibit a reduced plasma T concentration normally indicative of a female. Also, there is evidence that capture or other stressors can cause a reduction in plasma T concentration, again leading to possible errors in sex determination (Valverde *et al.*, 1996). Finally, these criteria should not be considered valid for use with reproductively active females.

A female bias was found in larger size classes, with  $\geq 40$  cm SCL *L. kempii* exhibiting a sex ratio of 1.7F:1.0M and all individuals greater than 60 cm SCL reported as female. Increasing female-bias with size may be a spatial artifact, explained by subadult to adult male behavioral differences that render them more difficult to capture using methods deployed in this study, as well as, less likely to strand. Stranding records exhibit a pattern similar to that of captured *L. kempii* and suggests that strandings and the capture lot used in this study may not be entirely representative of the at-large population.

If one assumes that the capture lot is representative of the population, the overall sex ratio reported for *L. kempii* in this study, 1.3F:1M, would apply. However, it is suggested that pubescent to adult male *L. kempii* maintain larger home ranges and remain farther from shore than do their female counterparts. Analysis of juvenile turtles captured during this study (circa 20 - 40 cm SCL) from which a blood sample was obtained (n = 149) suggests that the wild *L. kempii* population exhibits a primary sex ratio of near 1F:1M.

These results and a thorough review of scientific literature were used to develop a preliminary population model for *L. kempii*. The final model was used to assess the impact of various population sex ratio values upon Kemp's ridley sea turtle demography. Two scenarios were tested using the model. One in which proportion of adult males in the population have no effect on hatchling production and another where productivity is a function of the relative abundance of adult males.

Under the first scenario, the model suggests that a greater proportion of females in the population dramatically enhances hatchling production, yielding a 271% increase in predicted nests after 50 years with a sex ratio of 3F:1M and an 81% decrease with 1F:3M. The second scenario, in which reproductive output is a function of male availability, was approximated by assuming a linear relationship between increasing proportion of males and reproductive success. Strong female bias (3F:1M) resulted in a 20% decrease in the population while a male bias (1F:3M) yielded a 62% decrease. The greatest rate of reproductive return was achieved with a sex ratio of 1.28F:1M (56.2% female).

## **Conclusions**

Assumptions regarding sea turtle sex ratios have serious implications for both population modelers and managers. For example, the idea of using sex ratio

manipulation to “jump start” declining turtle populations has great appeal, but what would the impact be on the reproductive ecology of a species. Approximation of a fecundity effect in the model for this study attempts to simulate a scenario in which sperm storage, multiple paternity and/or sperm competition might play a role in fertility and population persistence. Under this scenario the relative abundance of adult males is critical to reproductive output, suggesting that there is an “optimum” sex ratio.

Factors related to male-mediated fecundity would only be effective when the number of reproductive males is sufficient to facilitate multiple inseminations of individual females. Sugg and Chesser (1994) have demonstrated the importance of breeding structure on gene diversity in natural and captive populations. They reported that multiple paternity increases the effective population size above that expected from polygyny or monogamy. As the number of mating males decreases, the impact of multiple paternity also decreases.

This suggests that multiple paternity and sperm competition may be important in the persistence of populations and cautions against manipulating the sex ratio of turtle populations to produce an excess of females. Model results from this study support the need for caution and further suggest that there is a point of diminishing returns beyond which increasing the proportion of adult females in the population no longer provides increased productivity. Without male-mediated fecundity there is little doubt that as the number of reproductive females increases hatchling production will also increase (Fig. 34). However, it is not clear that this is the case, and there is much evidence to the contrary.

Under the assumptions provided, the model specifies an adult sex ratio of 1.28F:1M (56.2% female) to optimize population growth (Fig. 34). The exact value of this “optimum” sex ratio depends greatly on the relationship between adult sex ratio and reproductive output (Fig. 23). Finding the actual “optimum” sex ratio would be

contingent upon quantifying this relationship. In addition, the fecundity effect in the model in no way addresses the potential for increased genetic fitness that might be gained from multiple paternity, only a potential increase in fecundity.

Potential problems are exacerbated when it comes to *L. kempii* as the sex ratio present in the population today may not represent a “natural” sex ratio. The majority of Kemp’s ridley nests have been transplanted to corrals since 1978 (Márquez, 1994). It is likely that most extant *L. kempii* came from transplanted clutches, and it is uncertain what effect nest transplantation has had on population sex ratio. Although temperature has been monitored both in the corrals and on the Mexican nesting beaches (Márquez, 1982b, 1983, 1985; Márquez *et al.*, 1986, 1987, 1989), it is not clear these data have been analyzed to fully quantify sex ratio effects related to incubation temperature. Certainly they have received no critical review. The big question is, does the *L. kempii* sex ratio presented herein reflect a natural sex ratio?

Limited data have been presented indicating that the beach at Rancho Nuevo offers a range of incubation temperatures at, above, and below the pivotal sex ratio temperature depending upon distance from the water, season, and rainfall (Standora and Spotila, 1985; Shaver *et al.*, 1988). Unfortunately, there is a paucity of published data comparing thermal profiles of transplanted nests versus those left *in situ*. The NMFS headstart experiment represents an extreme example of the possible effects of transplantation in the predominant production of male hatchlings before the pivotal incubation temperature was discovered and integrated into experimental protocol (Wibbels *et al.*, 1989). It is possible that transplanted nests are exposed to less temperature variation during incubation because most found nests are grouped together into relatively small areas. Such a change in the thermal profile of a nest may have unknown consequences for the sex ratio produced by that nest (Mrosovsky and Yntema, 1980).

These factors underscore the importance, not only of determining the existing population sex ratio for *L. kempii*, but also whether current population structure represents a natural state. Have ongoing conservation practices influenced population sex ratios, and if so, in what way? The population model presented in this study can serve as an invaluable tool in answering this question, providing a mechanism through which the population impact of *in situ* and natural nest sex ratios can be compared. Ongoing efforts to determine primary sex ratio production at the nesting beach via non-lethal sex determination in hatchlings is critical to these efforts (Wibbels, 1998; Wibbels and Geis, 1999).

Aside from sex ratio manipulation, the model developed in this study can be used to identify other areas of potential concern and serve as a useful tool in evaluating additional management strategies. Sensitivity analysis indicated that reduction of pelagic post-hatchling mortality had the single greatest impact on the population model, increasing nest production by 301% over baseline after 50 years (Table 6). Unfortunately protecting pelagic post-hatchling *L. kempii* is not a likely option in the foreseeable future as little is known about their habits or habitats. Reduction of juvenile and subadult mortality in the model also generated large increases in number of nest and has profound implications for current TED regulations and efforts underway to establish protected areas for the species.

An important question in this regard is, if TED regulations are working to protect sea turtles at-sea why have stranding rates not declined since their inception? Subadult, and certainly juvenile, *L. kempii* should be beneficiaries of existing TED regulations but stranding rates have continued to increase (Shaver, 1995). One answer is that stranding rates have remained the same or decreased, but, as the population recovers, there are simply more turtles and hence more strandings.

However, this question deserves a closer look, both for the long-term health

of the population and to prevent increasing numbers of dead turtles from washing up on shore. A better understanding of *L. kempii* distribution in time and space and across all life-history stages is critical. Such information would be invaluable in assessing which life-history stages are most likely to be impacted by trawl fisheries, examining when and where turtles occur in relation to fishing effort.

Similarly, knowledge of *L. kempii* distribution is important for identifying and protecting critical habitat for the species. A campaign is underway to establish a protected area for Kemp's ridley off of the Padre Island National Seashore (PINS) during the nesting season (Shore, 2000). These efforts are aimed at protecting the recent influx of new nesters to the PINS in hopes of fostering the establishment of a second nesting beach for *L. kempii*. It is also argued that this area would provide a protected swimway for adults as they migrate to and from their Mexican nesting beaches along the south Texas coast.

Identifying life-history stage and sex specific temporal and spatial distribution for *L. kempii* could aid in identifying the boundaries of such protected areas as well as other candidate areas. The population model presented here, and others like it, provide an invaluable tool in evaluating the potential impact of these conservation strategies. Within its given working assumptions, managers have the power to review and refine existing management practices and regulatory framework, and more accurately assess the potential impact of proposed conservation measures.

### Future Research Needs

The population model developed as part of this study was completed through extensive review of all available information related to sea turtle population modeling and the Kemp's ridley sea turtle. As a result of these efforts the following research needs were identified:

- Further refine the primary sex ratio for use in population modeling by examining sex in *L. kempii* hatchlings on the nesting beach.
- Fully describe the *in situ* nesting environment for *L. kempii* relative to that of nests transplanted to protective "corrals".
- Quantify the relationship between adult sex ratio and reproductive output to more accurately quantify the "optimum" sex ratio for *L. kempii*.
- Current conservation efforts on the nesting beach should be reviewed to ensure they are not adversely affecting primary sex ratio production with regard to the "natural" or "optimum" sex ratio.
- The distribution and habitat utilization of *L. kempii* in time and space should be investigated. In particular, effort should be focused on subadult (circa 40 - 60 cm SCL) male *L. kempii* to determine why they appear in lesser numbers than same size females in both capture and stranding data. Also determine to what extent this difference is a function of differential mortality and/or habitat utilization.
- Information regarding the distribution of *L. kempii* should be analyzed to aid in identifying potential protected areas. In particular, these data should be compared with the spatial and temporal distribution of trawl fishing to refine existing TED regulations and more accurately reflect potential conflict with sea turtles.
- Investigate the need for development of life-history stage and/or sex specific survivorship rates for more accurate population modeling.

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## APPENDICES

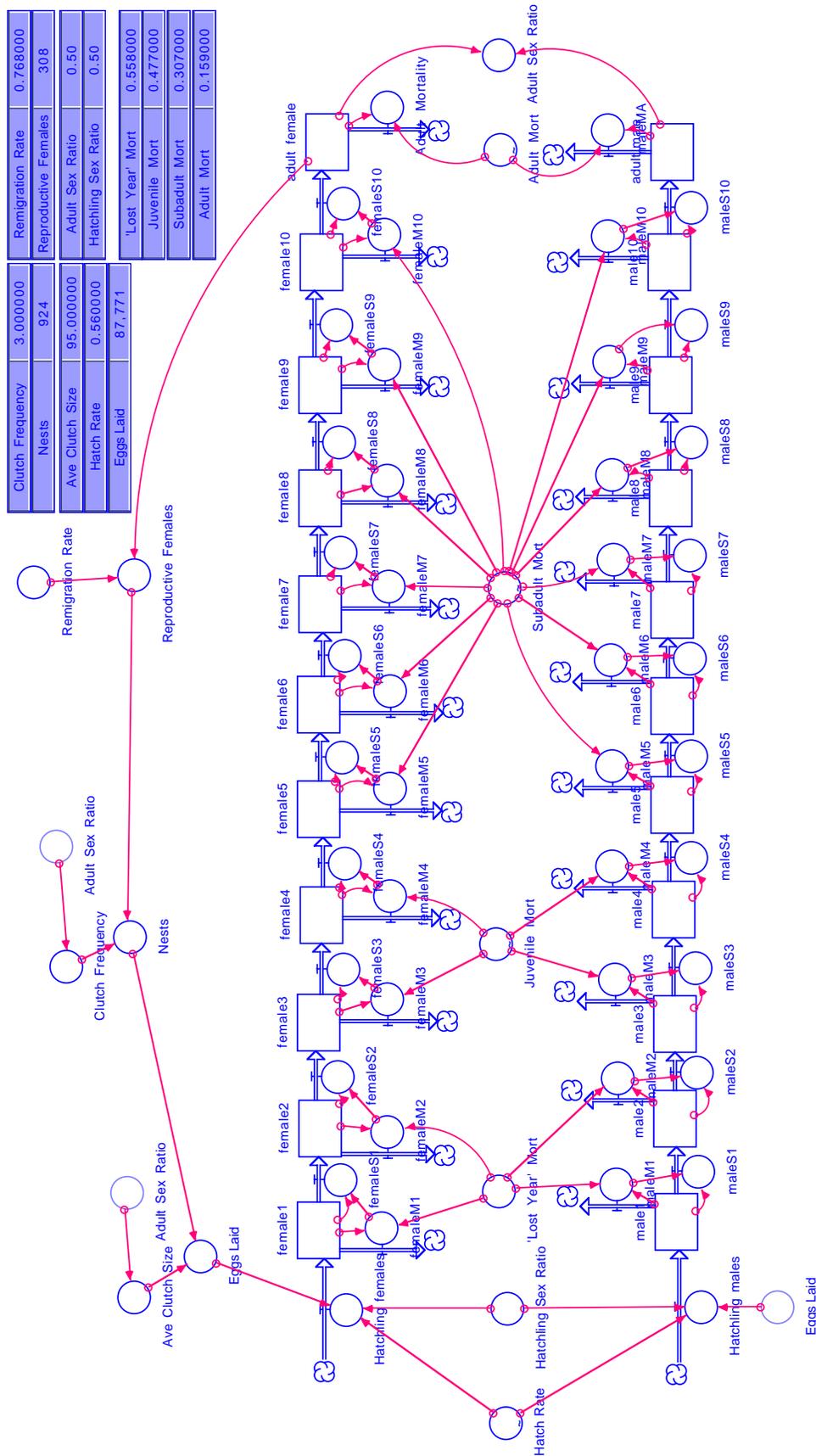
## APPENDIX A

Kemp's ridley capture /recapture data from the literature and this study used to calculate von Bertalanffy growth equation. Shaded entries were excluded from analyses.

<b>ID #</b>	<b>Capture Size (SCL cm)</b>	<b>Recapture Size (SCL cm)</b>	<b>Time-at-Large (years)</b>	<b>Growth (cm/yr)</b>	<b>Source</b>
A3857	68.0	62.5	0.19	-28.90	Marquez, 1994
K0003	71.5	68.5	0.21	-14.30	Marquez, 1994
A1279	65.0	64.0	0.08	-12.50	Marquez, 1994
J1028	65.0	57.5	0.97	-7.73	Marquez, 1994
G4888	63.5	63.0	0.13	-3.85	Marquez, 1994
SP95-7-8	68.5	64.0	1.17	-3.85	This study - nested
C94-7-2	68.0	64.6	1.14	-2.99	This study - nested
G9970	74.2	66.0	2.83	-2.90	Marquez, 1994
G0366	14.4	14.0	0.14	-2.86	McVey/Wibbels
A1012	68.0	65.5	1.11	-2.25	Marquez, 1972
G0467	13.0	12.7	0.15	-2.00	McVey/Wibbels
T0744	68.5	67.0	1.17	-1.28	Marquez, 1994
A1010	65.0	64.0	1.01	-0.99	Marquez, 1972
A1280	66.0	65.0	1.07	-0.93	Marquez, 1972
A1263	62.0	61.5	1.02	-0.49	Marquez, 1972
F0605	64.0	64.0	0.09	0.00	Marquez, 1994
G4708	69.5	69.5	0.25	0.00	Marquez, 1994
K0140	70.0	70.0	2.47	0.00	Marquez, 1994
C17797	71.0	71.0	3.87	0.00	Marquez, 1994
A1329	66.0	66.0	5.68	0.00	Marquez, 1994
G4633	63.5	63.8	9.81	0.03	Marquez, 1994
A1145	66.5	67.0	2.92	0.17	Marquez, 1994
A1263	66.0	68.0	4.14	0.48	Marquez, 1994
A1251	63.5	64.0	0.98	0.51	Marquez, 1972
A1232	62.5	63.0	0.97	0.52	Marquez, 1972
A1252	65.0	66.0	1.03	0.97	Marquez, 1972
PL96-7-3	43.9	44.8	0.82	1.10	This study - recapture
C13108	62.7	65.0	1.94	1.19	Marquez, 1994
A1437	65.5	66.6	0.73	1.50	Marquez, 1972
A1184	61.5	64.5	1.68	1.79	Marquez, 1972
A1117	62.0	71.0	5.00	1.80	Chavez/Kaufman
A1116	65.0	69.0	2.00	2.00	Marquez, 1972
G0104	15.2	15.5	0.13	2.26	McVey/Wibbels
A1116	65.0	69.0	1.71	2.34	Marquez, 1994
SSJ058	28.1	30.8	1.07	2.51	This study - hook and line
SSK055	65.8	67.6	0.70	2.58	Head Start
A1260	61.0	64.0	1.02	2.93	Marquez, 1972
A4499	65.0	66.0	0.29	3.45	Marquez, 1994
T0590	65.0	71.0	1.52	3.95	Marquez, 1994
G2406	14.0	18.5	0.97	4.62	McVey/Wibbels
SP97-8-5	47.1	51.1	0.76	5.26	Recapture
A3868	65.0	76.0	2.08	5.29	Marquez, 1994
QQA183	17.0	55.1	6.86	5.55	This study - headstart
SP93-4-3	41.0	45.7	0.82	5.72	This study - recapture
SP96-7-12	38.6	43.6	0.87	5.74	This study - recapture
A4558	65.5	70.0	0.78	5.81	Marquez, 1972
A1002	61.0	67.5	1.07	6.05	Marquez, 1972
SP93-7-1	33.6	38.5	0.79	6.23	This study - recapture
G2697	13.8	21.5	1.14	6.75	McVey/Wibbels

ID #	Capture Size (cm SCL)	Recapture Size (cm SCL)	Time-at-Large (years)	Growth (cm/yr)	Source
A1071	65.0	69.0	0.59	6.78	Sweat
QQX067	15.3	49.7	5.01	6.87	This study - headstart
SP97-6-13	36.0	42.7	0.94	7.13	This study - recapture
QQA650	17.1	57.9	5.04	8.09	This study - headstart
SP97-5-16	32.9	33.4	0.06	8.33	This study - recapture
SSK119/120	33.9	41.6	0.89	8.67	This study - stranded
G0460	15.5	30.4	1.67	8.92	McVey/Wibbels
G0985	17.2	25.0	0.86	9.09	McVey/Wibbels
G0904	17.0	29.2	1.32	9.24	McVey/Wibbels
SP93-4-3	35.4	45.7	1.11	9.28	This study - recapture
SP97-5-2	28.0	28.7	0.07	9.46	This study - recapture
SSH495	19.8	26.9	0.75	9.49	This study - headstart
G2667	15.0	25.5	1.09	9.63	McVey/Wibbels
QQL196	17.4	28.0	1.09	9.70	This study - headstart
SP93-7-24	22.2	24.0	0.18	9.95	This study - recapture
G4841	64.5	70.0	0.54	10.19	Marquez, 1994
QQL449	17.6	47.8	2.94	10.27	This study - headstart
G0190	15.4	30.5	1.46	10.34	McVey/Wibbels
SP97-5-18	29.1	39.7	1.00	10.60	This study - recapture
QQX835	14.8	36.7	2.05	10.67	This study - headstart
G2386	15.4	26.0	0.97	10.87	McVey/Wibbels
QQX732	15.3	38.0	2.04	11.11	This study - headstart
QQL137	17.3	39.6	1.96	11.35	This study - headstart
QQX073	15.3	26.7	0.98	11.59	This study - headstart
QQW143	14.8	39.5	2.13	11.62	This study - headstart
G0914	15.2	26.3	0.93	12.00	McVey/Wibbels
SSD816	16.5	30.1	1.13	12.05	This study - headstart
QQW470	14.1	26.3	1.00	12.17	This study - headstart
QQX028	14.9	26.9	0.98	12.20	This study - headstart
SSD880	16.4	33.2	1.34	12.51	This study - headstart
QQW451	13.3	26.1	1.00	12.85	This study - headstart
QQW781	15.3	30.5	1.14	13.30	This study - headstart
QQX641	14.4	31.0	1.24	13.35	This study - headstart
QQX804	15.0	32.5	1.23	14.26	This study - headstart
SSD402	15.4	32.0	1.13	14.71	This study - headstart
SP93-7-22	30.5	32.0	0.10	14.80	This study - recapture
SSD317	16.6	35.1	1.23	15.01	This study - headstart
QQW335	14.5	31.8	1.15	15.11	This study - headstart
QQW422	15.3	34.1	1.23	15.32	This study - headstart
QQW343	14.6	29.7	0.98	15.35	This study - headstart
QQX265	14.2	30.8	1.07	15.46	This study - headstart
SP93-7-2	46.8	48.5	0.11	15.51	This study - recapture
SP93-7-21	22.6	24.5	0.12	15.76	This study - recapture
QQW363	15.2	30.7	0.94	16.54	This study - headstart
A4515	65.0	67.0	0.12	16.67	Marquez, 1994
SSC584	16.5	40.0	1.34	17.51	This study - headstart
SP93-4-3	35.4	41.0	0.29	19.47	This study - recapture
G0618	18.4	21.4	0.14	21.43	McVey/Wibbels
C17180	62.9	66.8	0.13	30.00	Marquez, 1994
A4508	65.0	70.0	0.08	62.50	Marquez, 1994
C17199	67.0	68.5	0.02	75.00	Marquez, 1994

### APPENDIX B



Conceptual Kemp's ridley sea turtle population model as represented in Stella Research Software version 5.1.1. Some initial model variables are shown in the shaded boxes.

## VITA

### **Michael Scott Coyne**

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Michael Scott Coyne was born on March 8th, 1967 in Lafayette, Indiana to Dr. George W. and Laura J. Coyne. His family entered the Peace Corps from 1971 - 76 serving in Cape Coast, Ghana. He moved to Winter Park, Florida in 1979 with his family, where he graduated from Winter Park High School in May, 1985. He entered Valencia Community College (Orlando, Florida) that fall, then graduated in May 1987 with an Associate of Arts degree. He transferred to Drexel University (Philadelphia, Pennsylvania) to pursue a Bachelor of Science degree, transferring again to the University of Florida (Gainesville, Florida) in May 1989, graduating with a Bachelor of Science degree in Zoology during May 1990. While an undergraduate he gained laboratory experience working for the U.S. Department of Agriculture (Germantown, Pennsylvania), the Department of Bioscience and Biotechnology of Drexel University, and the Department of Gastroenterology at Shands Hospital, University of Florida. He entered graduate school in the Department of Wildlife and Fisheries Sciences at Texas A&M University in September 1990 where he received a Master of Science degree in 1994. As a graduate student he served as a teaching assistant in the freshman biology program in College Station and Galveston and in the marine biology program at Galveston. In addition, he participated in projects involving sea turtles as a research assistant at Texas A&M University at Galveston, presenting the results at scientific meetings. Currently he is working for the National Ocean Service's Biogeography Program as a marine biologist and serves as the online coordinator for the Marine Turtle Newsletter. Michael is married to Lynette Christine Coyne and has a beautiful daughter Isabel Christean.