

Trends in Kemp's Ridley Sea Turtle (*Lepidochelys kempii*) Relative Abundance, Distribution, and Size Composition in Nearshore Waters of the Northwestern Gulf of Mexico

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Long-term monitoring of in-water life history stages of the critically endangered Kemp's ridley sea turtle (*Lepidochelys kempii*) is essential for management because it generates information on the species' at-sea abundance, size composition, distribution, and habitat requirements. We documented trends in Kemp's ridley size, relative abundance, and distribution using entanglement netting surveys at three study areas adjacent to tidal passes in the northwestern Gulf of Mexico (NWGOM) during intermittent sampling periods from 1991 to 2013. A total of 656 Kemp's ridley sea turtles were captured ranging in size from 19.5 to 66.3 cm straight carapace length (SCL) (mean = 35.0 cm SCL). The dominance of juveniles (25–40 cm SCL) captured during sampling suggests the nearshore waters of the NWGOM are an important developmental foraging ground for Kemp's ridley. Characterization of Kemp's ridley long-term relative abundance reveals a generally stable trend in catch-per-unit-effort (CPUE) across all study areas combined. Based on the increasing trend in the number of hatchlings released from the species' primary nesting beach, Rancho Nuevo, Mexico, since the early 1990s, the lack of a corresponding overall increase in juvenile abundance at nearshore sampling locations is puzzling. This disparity is most likely an artifact of the present study's sampling design, but could also indicate shifts in Kemp's ridley recruitment away from the NWGOM. While conservation efforts have contributed to this species' overall growth since the 1980s, as measured by the increasing number of nests, recent declines in this rate of increase are a concern and call for a more comprehensive approach to managing Kemp's ridley recovery efforts.

INTRODUCTION

The Kemp's ridley sea turtle, *Lepidochelys kempii* (Garman 1880), is the smallest and most critically endangered sea turtle in the world (IUCN, 1996). In contrast to most other sea turtle species with circumglobal distributions, Kemp's ridley is primarily confined to the Gulf of Mexico (GOM) and United States east coast and has only one major nesting beach at Rancho Nuevo, Tamaulipas, Mexico (~ 23°N, 97°45'W). In-water captures, stranding surveys, and tracking studies indicate that benthic immature Kemp's ridley sea turtles, 20–60 cm straight carapace length (SCL), primarily inhabit nearshore waters, extending as far north as Massachusetts during summer (Hildebrand, 1982; Ogren, 1989; Márquez, 1994; Landry and Costa, 1999; TEWG, 2000; Seney and Landry, 2008). All neritic life stages, from juveniles through adults, can be found in the northern GOM from Texas to Florida at beachfront and tidal pass habitats, seasonally foraging on portunid crabs, particularly blue crabs (*Callinectes sapidus*) in the NWGOM (Shaver, 1991; Werner, 1994; Schmid,

1998), and shrimp bycatch. In-water capture statistics indicate the highest seasonal abundance of Kemp's ridleys in nearshore waters of the northern GOM occurs during April to August, followed by sharp declines from November through March as they move into deeper, warmer waters with the onset of cooler water temperatures (Renaud, 1995; Landry and Costa, 1999; Renaud and Williams, 2005).

Effective management and recovery of Kemp's ridley necessitates a greater understanding of factors influencing this species' in-water occurrence and survivorship (NRC, 2010). To date, most ridley conservation measures have focused on nesting beach protection and reduction of incidental capture in commercial shrimp trawls (NMFS et al., 2011). While these efforts have seemingly contributed to a 12–19% annual increase in nests at Rancho Nuevo since 1985 (Heppell et al., 2005; Crowder and Heppell, 2011; NMFS et al., 2011), recent declines in nesting activity in 2013–14 have raised concerns about the future status and continued recovery of this species (Caillouet, 2014). There is a lack of information on habitat requirements and

long-term trends in abundance and distribution of in-water life history stages. Furthermore, the connection between juvenile Kemp's ridley abundance in nearshore habitats and patterns of nesting activity at Rancho Nuevo has not been documented.

Tracking studies indicate that the northern GOM serves as important foraging grounds and a migratory corridor for adult female Kemp's ridleys (Renaud, 1995; Renaud and Williams, 2005; Shaver and Rubio, 2008; Shaver et al., 2013), but few studies have examined the habitat use patterns of benthic juveniles. Radio and sonic tracking of Kemp's ridleys released near Sabine Pass, TX, and Calcasieu Pass, LA, demonstrated that smaller individuals (< 50 cm SCL) exhibit site fidelity to tidal passes, staying within 15 km of their release site, while larger Kemp's ridleys (> 50 cm SCL) are more migratory, traveling > 17 km from shore and as far as 2600 km after release (Renaud and Williams, 2005). More recent satellite tracks of benthic juvenile Kemp's ridleys taken incidentally on hook-and-line and in dredge relocation trawls since 2004 also reveal a strong attraction to shallow nearshore waters of the NWGOM (Seney and Landry, 2011). These juvenile Kemp's ridleys showed a tendency to enter tidal passes from Pass Cavallo to Calcasieu Pass, and some ventured into various bays, especially Lavaca-Matagorda Bay, Galveston Bay, Sabine Lake, and Lake Calcasieu.

Although previous research has been valuable in determining Kemp's ridleys' overall range of occurrence and short-term habitat use, few studies have examined long-term abundance trends or distribution patterns. Long-term monitoring of in-water life history stages is essential for the management of Kemp's ridley because it generates information about at-sea abundance, size composition, distribution, and habitat requirements. These data are necessary for models used to evaluate population status and detect changes in abundance and structure that can impact future population growth and reproductive success (Epperly, 2000; NRC, 2010). Crouse et al. (1987) used a Lefkovich stage-based model to show that increased survival of juveniles and subadults was more effective in promoting loggerhead (*Caretta caretta*) population growth than was protection of eggs and hatchlings due to the higher reproductive value of large immature turtles. An age-based, deterministic model of Kemp's ridley population dynamics developed by the Turtle Expert Working Group

(TEWG 2000) was questionable, due in part to insufficient data on juvenile Kemp's ridley survivorship. More recently, Gallaway et al. (2013) modified the TEWG model to create a stock assessment model that incorporated shrimping mortality estimates, incremental growth of tagged and recaptured Kemp's ridleys, and length frequency data from stranded turtles, as well as a 2010 nest reduction multiplier that was necessary to fit the data for 2010 and beyond. These adjustments resulted in the model estimating the total population of age 2 + Kemp's ridleys to have been 248,307 in 2012.

Absolute abundance of in-water sea turtle life history stages is difficult to quantify due to the vast size of oceans and the inability to sample all habitats and individuals. Indices such as catch-per-unit-effort (CPUE) and mark-recapture analyses may be used to evaluate changes in relative abundance over time or among habitats/locations (Epperly, 2000; Bjorndal et al., 2005; Metz and Landry, 2013). However, it is important to note that changes in CPUE (or lack thereof) may not reflect actual changes in population abundance, especially when the catchability coefficient (i.e., proportion of the stock captured by one unit of effort) is unknown or unquantified (Maunder et al., 2006). An assumption of constant catchability is often made when using CPUE as a measure of relative abundance, but this assumption is commonly violated due to multiple factors that influence catchability, such as behavior and distribution of target species, changes in fishing effort or efficiency, and environmental factors (Katsanevakis et al., 2012). As such, these factors must be considered when interpreting trends in CPUE data.

To provide long-term data on in-water life history stages, we documented trends in Kemp's ridley size, CPUE, and distribution via entanglement netting surveys at nearshore and inshore locations adjacent to tidal passes in the NWGOM during intermittent sampling periods from 1991 to 2013. Additionally, we examined the relationship between juvenile Kemp's ridley occurrence in the NWGOM and nesting productivity at Rancho Nuevo to better understand nearshore recruitment dynamics. We hypothesized that annual CPUE of juvenile neritic Kemp's ridleys (20–60 cm SCL) at study areas in the NWGOM is correlated to the number of hatchlings released from Rancho Nuevo since 1990 and therefore would show a significant increasing trend during the study period.

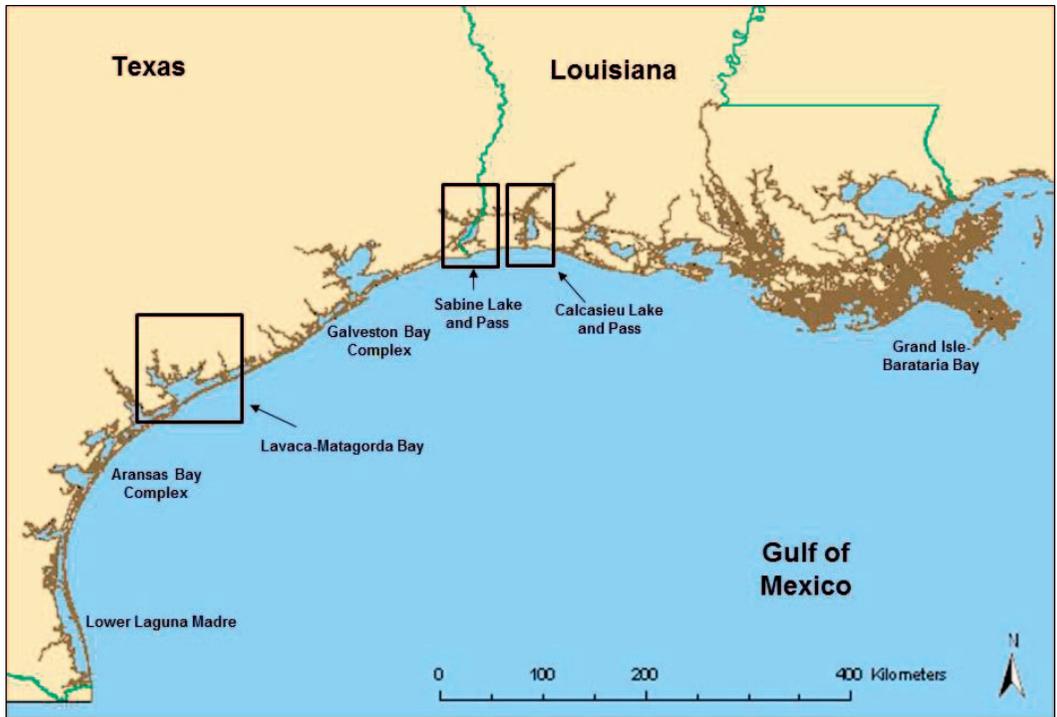


Fig. 1. Map of historical netting areas. Locations within the black boxes are the primary focus of this study.

METHODS

Study areas.—We conducted in-water sampling from 1991 to 2013 along the Louisiana and Texas coasts at nearshore and inshore locations from Grand Isle, LA, to the lower Laguna Madre near Port Isabel, TX (Fig. 1). The present study focuses on the data collected at Calcasieu Pass/Lake (1993–95, 1998–2002, 2011, 2013), Sabine Pass/Lake (1993–2002, 2006, 2008, 2013), and Lavaca-Matagorda Bay (1996, 2001–02, 2006–07, 2012–13), based on greater number of years sampled at these locations and the historical distribution of Kemp's ridleys in the NWGOM (Manzella and Williams, 1992; Landry and Costa, 1999).

Sabine Pass forms the southernmost border between Texas and Louisiana, with Calcasieu Pass located 46.3 km to the east in Cameron Parish, LA. Sabine Pass is bordered on the east and west by 5.6 km long granite jetties, near which sea turtle monitoring stations have been established at inshore and beachfront locations. Beachfront stations were located within 1 km of the jetties and 300–1500 m offshore, in waters 0.6 to 3.0 m deep with soft muddy/clay substrate and compacted sandy/mud bottom. The nearshore topography and position of jetties at Calcasieu

Pass were similar to those at Sabine Pass, with the exception of shorter jetties (approximately 3 km) protecting the pass. Sampling stations at Calcasieu Pass mirrored those at Sabine Pass in terms of number, designation, and relative location. Calcasieu Pass data for 1999 and 2000 also include ridley captures from beachfront sites adjacent to the mouth of the Mermentau River, LA, located ~ 22 km to the east. Lavaca-Matagorda Bay is located ~ 280 km to the SW of Sabine Pass, near Port Lavaca, TX. All netting sites within this study area were located inshore with bottom types ranging from seagrass beds and sand to soft, muddy clay and shell hash further inshore.

Sea turtle capture.—We captured turtles in-water during daytime sets using 0.0914 km long entanglement nets 2.7–4.9 m deep with 12.7–25.4 cm bar mesh of No. 9 twisted nylon. Water depth and current dictated net type used at a particular station, with all capture effort restricted to depths < 3 m. Netting effort at all stations consisted of two to four nets set in line or perpendicular to one another for 6–12 hr/d. Typically, one boat with one to four observers was responsible for monitoring two nets that were checked for sea turtles and bycatch every 20

min, or more frequently as splashes or other signs of potential capture dictated. We attached pinger devices (Fumunda Marine, Queensland, Australia), which emitted high-frequency sounds, to nets to reduce incidental capture of bottlenose dolphins (*Tursiops truncatus*). We conducted entanglement netting surveys during April–October each year, but not all months were sampled every year, nor were all sites sampled each year. Although 3 d of sampling effort was our target for each study area during respective visits, weather conditions and equipment problems at times prevented netting in a particular area or month. Additionally, lack of or insufficient funding during some periods prohibited sampling and resulted in data gaps. This in turn resulted in unequal and/or missing data that precluded monthly and seasonal comparisons. Nevertheless, we conducted enough monthly sampling to allow for annual comparisons.

We measured the straight carapace length (SCL) in centimeters of all sea turtles captured and examined them for flipper tags/scars, Passive Integrated Transponder (PIT) tags, living tags, and coded wire tags (the latter two tag types denoting head-start Kemp’s ridleys, which were part of a conservation program from 1978 to 1992 to reduce mortality by rearing turtles in captivity from hatchlings to 1–2 yr of age before release into the GOM). Untagged sea turtles received an inconel style 681 tag, issued by the Cooperative Marine Turtle Tagging Program (CMTTP) at the Archie Carr Center for Sea Turtle Research (ACCSTR) at the University of Florida at Gainesville, affixed to the trailing edge of each front flipper and a PIT tag injected in the dorsal surface of the right front flipper prior to release at their capture location.

Data analyses.—We conducted a two-way analysis of variance (ANOVA) to analyze differences in netting effort between the three study areas and two blocks of years (1993–2002 vs 2003–13). We converted the entanglement netting capture data into a monthly or annual measure of catch-per-unit-effort (turtles/km hr) using the following equation:

$$\frac{N_T}{T(L)}$$

with N_T = total number of Kemp’s ridleys captured in entanglement net(s), T = total soak time (hr) of all net(s) deployed, and L = length of one net (0.0914 km). We then log trans-

formed [$\log(\text{CPUE} + 1)$] monthly and annual Kemp’s ridley CPUE values to approximate a normal distribution. We used a one-way ANOVA and Student’s t -test to identify differences in log transformed Kemp’s ridley CPUE and size between years at Sabine Pass, Calcasieu Pass, and Lavaca-Matagorda Bay. If ANOVA results showed significant differences between variables, we conducted post hoc comparisons using the least significant difference (LSD) test. We analyzed trends in annual mean CPUE and size with the null hypothesis that there was no significant trend (i.e., the slope of the regression line was equal to zero). If no trend was detected, we quantified the power of the regression analysis using the R statistical package “pwr” (R Core Team, 2013).

The number of Kemp’s ridley hatchlings released each year was obtained from annual reports prepared by the binational team of Mexican and U.S. researchers monitoring the nesting beaches in and near Rancho Nuevo (Burchfield and Peña, 2013). We analyzed the relationship between Kemp’s ridley annual mean CPUE for individuals sized 30–40 cm and the number of hatchlings released from Rancho Nuevo during 1990–2013 (plotted with a 3-yr lag to account for the pelagic stage and estimated age of most Kemp’s ridleys encountered) using least squares linear regression analyses, with both variables log transformed [$\log(\text{No. hatchlings})$; $\log(\text{CPUE} + 1)$] to normalize the data.

We examined monthly and annual Kemp’s ridley CPUE data for any autocorrelation between values that could confound regression analysis results. Neither monthly nor annual ridley CPUE was significantly autocorrelated (all Durbin–Watson test statistics ~ 2.0 , over 72 lags for monthly CPUE and nine lags for annual mean CPUE). Mean values are expressed as mean \pm SE. We conducted all statistical analyses using an α level of 0.05 in Microsoft Excel and SPSS 14.0 statistical software packages.

RESULTS

Netting effort.—A total of 1118.2 km hr of netting effort was expended in this study but was inconsistent across years and study areas (Table 1). Significantly less effort was expended in more recent years (2003–13) compared with 1993–2002 (two-way ANOVA, $F = 48.49$; $df = 1, 57$; $P < 0.001$) and Lavaca-Matagorda Bay received less effort (two-way ANOVA, $F = 6.44$; $df = 2, 57$; $LSD P = 0.003$) than the other two study areas, which

TABLE 1. Annual entanglement netting effort (km hr) at the three study areas included in this study during April–October 1993–2013.

Year	Sabine Pass	Calcasieu Pass	Lavaca-Matagorda Bay	Totals
1993	116.70	19.98	—	134.53
1994	37.76	55.58	—	88.2
1995	51.62	24.55	—	73.15
1996	67.80	—	45.45	111.33
1997	88.45	—	—	82.57
1998	61.33	24.25	—	85.58
1999	56.48	108.21	—	164.69
2000	29.98	53.38	—	83.36
2001	33.37	61.82	15.3	110.48
2002	26.18	47.25	54.72	128.15
2006	5.62	—	7.93	13.55
2007	—	—	16.38	16.38
2008	4.85	—	—	4.85
2011	—	8.37	—	8.37
2012	—	—	2.23	2.23
2013	1.23	4.23	6.55	10.78
Totals	630.05	403.97	148.56	1118.2

were not significantly different from each other (LSD $P = 0.159$).

Sea turtle captures.—In total, we captured 656 Kemp's ridleys during this study, with 92.1% netted at the Sabine and Calcasieu study areas. Sampling at Sabine netted 370 Kemp's ridleys, with the majority captured prior to 1999 ($n = 345$ prior vs $n = 25$ after) and no Kemp's ridleys encountered in 2000, 2006, and 2008. Wild, head-start, and wild-recaptured counterparts accounted for 90%, 6%, and 4%, respectively, of all Kemp's ridleys captured during 1993–98; thereafter, the proportion of wild caught Kemp's ridleys increased to 96%. Overall, fewer Kemp's ridleys were captured at Calcasieu Pass ($n = 234$) during 1993–2002 (no sampling occurred in 1996 and 1997), though post-1998 capture totals exceeded those for Sabine Pass ($n = 168$ at Calcasieu vs $n = 25$ at Sabine). Percentage contribution among wild, head-start, and wild-recaptured Kemp's ridleys at Calcasieu Pass mirrored that at Sabine Pass, with a similar increase in the proportion of wild captures and a decrease in head-start and wild-recaptured Kemp's ridleys after 1998. Limited sampling effort at Calcasieu in 2011 and 2013 yielded three wild Kemp's ridleys in 2011 and no captures in 2013. Lavaca-Matagorda Bay was sampled intermittently from 1996 to 2013, with 52 Kemp's ridleys captured over all years combined. The majority of these captures were

wild Kemp's ridleys (94.2%), with no head-start recaptures in this study area.

Size composition.—Straight carapace length (cm) of Kemp's ridleys captured across all three study areas from 1993 to 2013 ranged from 19.5 to 66.3 cm (mean = 35.7 ± 0.35 cm, $n = 652$). Kemp's ridley size at Sabine Pass ranged from 19.5 to 64.0 cm SCL, while those captured at Calcasieu Pass and Lavaca-Matagorda Bay ranged from 22.4 to 66.3 cm and 24.5 to 58.4 cm SCL, respectively. Average size peaked in 1995 (39.9 ± 1.42 cm, $n = 48$) and 2000 (39.7 ± 1.86 cm, $n = 21$) and was lowest at Lavaca-Matagorda Bay in 2006 (30.8 ± 1.10 cm, $n = 8$), but annual means at all sites combined did not exhibit a significant trend (linear regression: slope = -0.287 , $R^2 = 0.23$, $P = 0.07$, $n = 15$). However, there appeared to be a reduction in the size range of Kemp's ridleys captured at Sabine Pass after 1998, with an absence of individuals larger than 55.0 cm.

There were significant differences in mean SCL between study areas over all years sampled (One-way ANOVA: $F = 4.32$, $P = 0.014$, $n = 652$). Kemp's ridleys were significantly larger at Calcasieu Pass (mean = 37.1 ± 0.62 cm, LSD $P = 0.006$, $n = 234$) than at Sabine Pass (mean = 35.0 ± 0.47 cm, $n = 366$), but Kemp's ridleys from Lavaca-Matagorda Bay were not significantly different in size than those from Sabine or Calcasieu Passes (mean = 34.9 ± 0.81 cm, LSD $P = 0.82$ and 0.08 , respectively, $n = 52$). A majority of all Kemp's ridleys captured during the study were between 25 and 40 cm SCL (70.6%), with the 30.0–34.5 cm size class having the greatest percentage frequency at all three study areas and larger individuals less common (Fig. 2a–c).

Trends in CPUE.—Mean annual Kemp's ridley CPUE at Sabine and Calcasieu Passes during 1993–2002 is shown in Figure 3a. Annual combined mean CPUE at Sabine and Calcasieu exhibited peaks in 1994 (0.98 ± 0.29 /km hr, $n = 12$), 1997 (0.93 ± 0.67 /km hr, $n = 6$), 1999 (0.69 ± 0.20 /km hr, $n = 10$), and 2002 (0.52 ± 0.12 /km hr, $n = 5$). Although Kemp's ridley catch rates at Sabine Pass peaked in 1994 (1.33 ± 0.39 /km hr, $n = 7$) and 1997 (1.33 ± 0.39 /km hr, $n = 7$), CPUE declined to 0 in 2000 (Fig. 3a). Thereafter, CPUE in Sabine Pass gradually increased through 2002 to near 1998 levels. Overall, the trend line for Kemp's ridley captures at Sabine Pass was negative and significantly different than zero (linear regression: slope = -0.104 , $R^2 = 0.57$, $P = 0.011$, $n = 10$), indicating

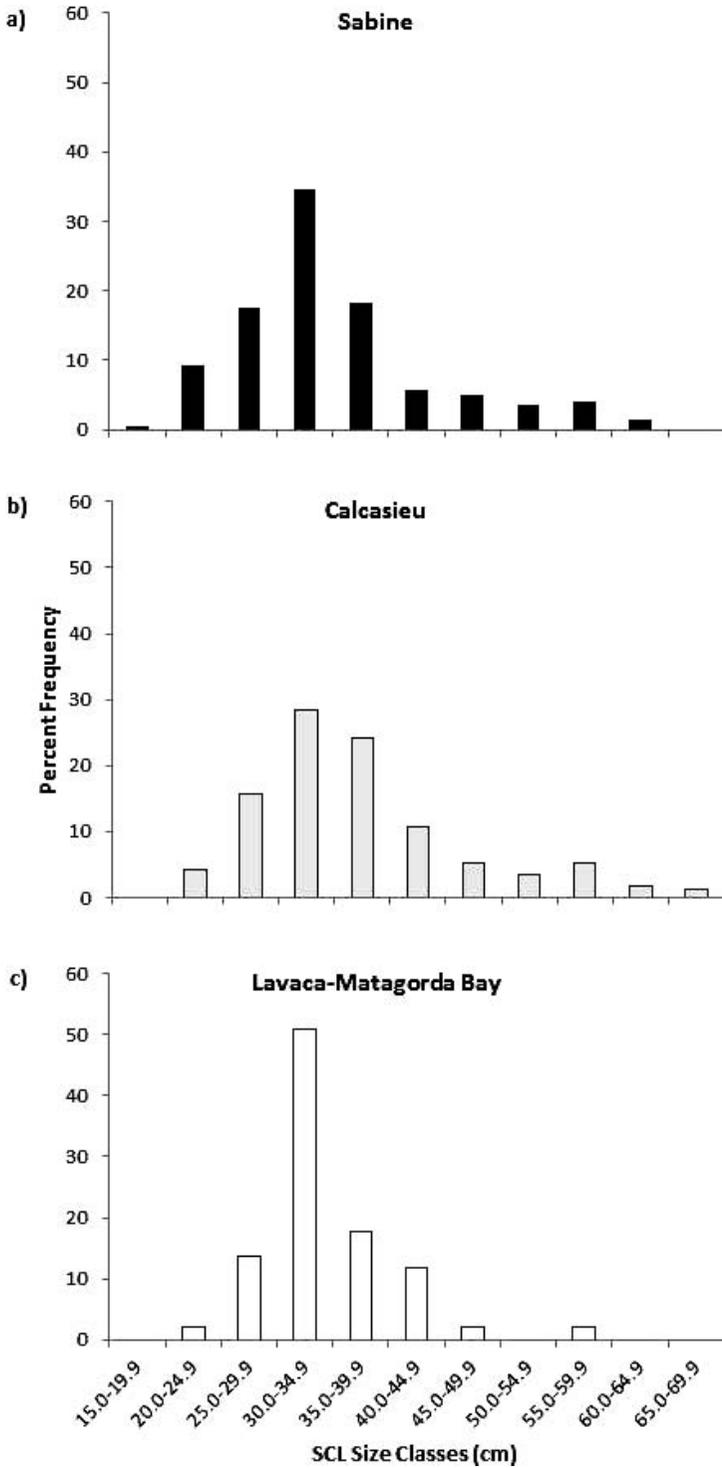


Fig. 2. Length frequency histograms of Kemp's ridleys netted at (a) Sabine (n = 366); (b) Calcasieu (n = 234); and (c) Lavaca-Matagorda Bay (n = 52) during April–October 1993–2013.

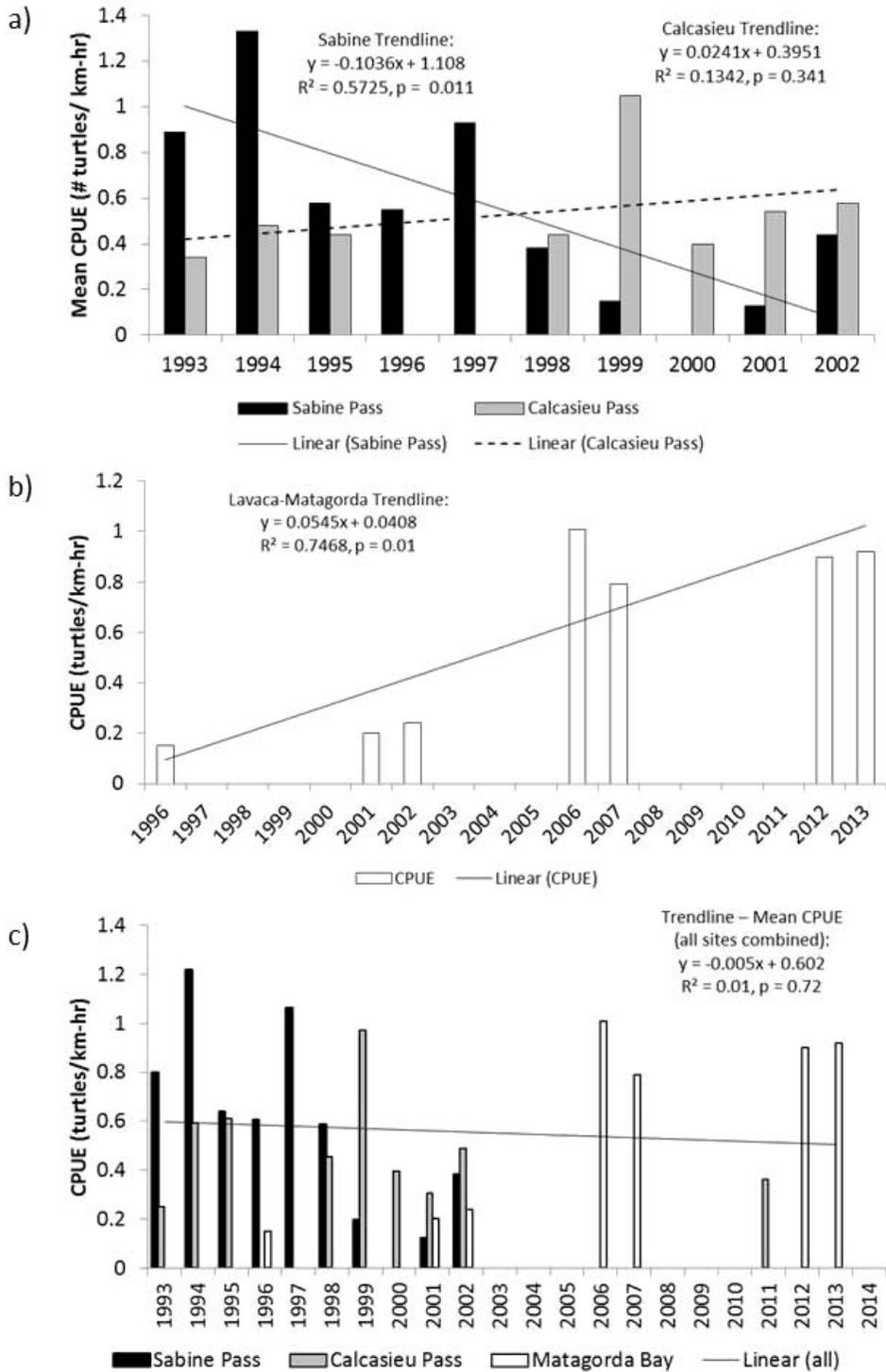


Fig. 3. Historical trends in Kemp’s ridley CPUE at (a) Sabine and Calcasieu (April–October 1993–2002; (b) Lavaca-Matagorda Bay (April–October 1996, 2001–02, 2006–07, 2012–13; and (c) all three study areas during April–October 1993–2013.

an annual decline in Kemp's ridley CPUE at this site from 1993 to 2002.

Annual Kemp's ridley CPUE at Calcasieu Pass was relatively stable across years (linear regression: slope = 0.026, $R^2 = 0.14$, $P = 0.341$, $n = 8$), with the exception of a major peak (1.05 ± 0.48 /km hr, $n = 6$) in 1999 (Fig. 3a). Even with this peak, there was no significant difference in CPUE between years (One-way ANOVA: $F = 1.20$, $P = 0.34$, $n = 34$). However, there appeared to be a shift in abundance and distribution between Sabine and Calcasieu Passes before and after 1997. Comparison of monthly CPUE values from these locations confirmed that significantly higher mean CPUE was observed at Sabine Pass prior to 1997 (Student's t-test: $t = 2.41$, $P = 0.022$, $n = 35$) and the converse thereafter ($t = -4.196$, $P < 0.00$, $n = 38$). Limited entanglement netting at Calcasieu Pass in 2011 resulted in Kemp's ridley CPUE of 0.36 turtles/km hr, which is similar to values observed in this location in 1993 and 2000.

Although Lavaca-Matagorda Bay was sampled more sporadically than the other two study areas, it provides another long-term dataset with more recent observations for evaluation of CPUE trends. Kemp's ridley annual CPUE has shown a significantly increasing trend within this bay system since 1996 (linear regression: slope = 0.054, $R^2 = 0.75$, $P = 0.01$, $n = 7$; Fig. 3b). However, when plotted with the overall annual CPUE values observed at Sabine and Calcasieu Passes, data from recent years in Lavaca-Matagorda are not significantly different than peak years previously recorded at the other two study areas (Fig. 3c). The slope of a line fitted to these data over all years is not significantly different than 0 (linear regression: slope = -0.005 , $R^2 = 0.01$, $P = 0.72$, $n = 15$).

Relationship of nearshore Kemp's ridley CPUE to hatchlings.—As the number of Kemp's ridley hatchlings released from the Rancho Nuevo nesting beach increased exponentially from 1990 to 2013 (Nonlinear regression: $y = 63692e^{0.1282x}$, $R^2 = 0.92$, $P < 0.001$, $n = 24$), Kemp's ridley CPUE at our study areas in the NWGOM remained relatively stable. Regression analysis of log transformed annual number of hatchlings released from Rancho Nuevo and log transformed CPUE for 30–40 cm Kemp's ridleys in developmental habitats (plotted with a 3-yr lag to account for the pelagic stage and estimated age of nearshore Kemp's ridleys based on SCL)

yielded no significant relationship (linear regression: $R^2 = 0.21$, $P = 0.09$, $n = 14$).

DISCUSSION

Small benthic juveniles (25–40 cm SCL) dominated the nearshore Kemp's ridley assemblage ($\sim 70\%$), indicating this area is an important developmental foraging habitat. Age estimates of wild Kemp's ridleys using skeletochronological data and the von Bertalanffy growth equation suggest the majority of Kemp's ridleys in this size range are 2–4 yr old (Zug et al., 1997; TEWG, 2000). These size/age observations are consistent with Kemp's ridleys that have recently transitioned to nearshore feeding grounds from the pelagic stage (Ogren, 1989; Manzella and Williams, 1992; Landry and Costa, 1999). The slightly larger size of Kemp's ridleys at Calcasieu Pass compared with those at Sabine Pass and Lavaca-Matagorda Bay to the west may be an artifact of the intermittent and unequal sampling effort across sites but may also indicate that Kemp's ridleys recruit to coastal areas in the western GOM at a smaller size/younger age than to the east. Simulated particle tracking to estimate dispersal of hatchlings from Kemp's ridley nesting beaches to coastal areas of the GOM and Atlantic by Putman et al. (2013) supports this possibility. Their results indicated that mean particle age generally increased from west to east across the GOM. Similarly, Witherington et al. (2012) found that the modal size of Kemp's ridleys (250 mm) associated with sargassum mats in the eastern GOM was skewed negatively (favoring larger turtles) and was similar to the lower size observed within nearby coastal-neritic habitats, suggesting these turtles were on the cusp of neritic recruitment. Alternatively, the higher frequency of large Kemp's ridleys at Calcasieu Pass may be due to larger immatures and adult females (> 60 cm) using foraging habitat in shallow, coastal waters off the Louisiana coast for reproductive conditioning (Renaud and Williams, 2005; Seney and Landry, 2008; Shaver and Rubio, 2008; Shaver et al., 2013).

Based on the increasing trend in the number of Kemp's ridley hatchlings released from Rancho Nuevo since the early 1990s (Crowder and Heppell, 2011), we expected a corresponding increase in benthic juvenile CPUE, with a time lag to allow for the pelagic stage. However, there was no significant relationship between the two, and characterization of Kemp's ridley long-

term abundance and distribution in nearshore waters of the NWGOM revealed a relatively stable trend in Kemp's ridley CPUE but with considerable variation around the trend line and low statistical power (0.15). Given this low statistical power and our lack of information on catchability of Kemp's ridleys during the span of this study, it is difficult to draw strong conclusions regarding overall trends in the relative abundance of benthic stage Kemp's ridleys in the NWGOM, especially in relation to nesting productivity. Although certain factors affecting catchability were standardized, such as established netting stations and size/type of nets used, effort was not consistent across years and locations. The significantly lower effort expended at study areas since 2003 compared with 1993–2002 may explain the lack of discernable trends, if there is a certain threshold of netting effort that must be employed. This same methodology, with variable effort, was used by the authors to reveal a significant exponential increase in annual CPUE of green turtles (*Chelonia mydas*) in the lower Laguna Madre, Texas, from 1991 to 2010, most likely due to elevated nesting productivity at beaches in Mexico, Florida, the Caribbean, and the western Atlantic (Metz and Landry, 2013). However, dissimilarities in diet and habitat use of Kemp's ridleys and green turtles likely affect catchability of each species, and the level of effort needed to detect trends in relative abundance may be different. Thus, more consistent netting effort that is on par with levels expended during the 1993–2002 period should be implemented in future studies of benthic juvenile Kemp's ridleys in the NWGOM to adequately assess trends and make historical comparisons.

Lack of a corresponding increase in Kemp's ridley coastal juveniles in response to increasing numbers of hatchlings most likely was not due to a decline in survivorship of benthic life history stages during the study period, given the reduction in threats resulting from mandatory implementation of turtle excluder devices (TEDs) in 1994 and a decrease in GOM shrimping effort over the past 20 yr (Caillouet et al., 2008; Nance et al., 2010). Finkbeiner et al. (2011) estimated a greater than 60% reduction in sea turtle–shrimp fishery interactions and a more than 90% reduction in sea turtle mortality in the southeast United States and GOM shrimp fishery between 1990 and 2007. Models of Kemp's ridley population growth prior to 2009 yielded 12–19% increases in nests each year and

closely matched parameters observed at the Rancho Nuevo nesting beaches (Heppell et al., 2005; Crowder and Heppell, 2011; NMFS et al., 2011). However, there has been a change in the trajectory of Kemp's ridley growth since 2010, and reasons for this change are still being evaluated (Caillouet, 2011, 2014; Gallaway et al., 2013). The Deepwater Horizon oil spill in 2010 may have led to increased mortality of Kemp's ridley pelagic through neritic life history stages or decreased foraging quality of nearshore waters that affected reproductive condition and the remigration interval of adult females. Direct and indirect effects on long-term health of Kemp's ridleys and reproductive conditioning could result from acute and chronic exposure to oil in the environment or from indirect effects on foraging quality, namely, those affecting blue crab stocks in coastal waters of the GOM (Shigenaka et al., 2003). The short- and long-term impacts of the Deepwater Horizon Oil Spill in 2010 on sea turtle populations in the GOM are not yet known or understood and warrant further investigation.

Assuming Kemp's ridley mortality rates remained relatively constant during this study, another possible explanation for the absence of a relationship between CPUE trends in the NWGOM and increased hatchling productivity at Rancho Nuevo is that posthatchlings are recruiting in higher numbers to coastal locations outside of this region due to factors such as dispersal via ocean currents and coastal foraging quality. Distribution could be affected by ocean currents which carry posthatchlings from nesting beaches to benthic foraging grounds. Putman et al. (2013) simulated Kemp's ridley hatchling dispersal from beaches in Mexico and Texas by use of particle drift models. Although simulated dispersal of hatchlings resulted in more than half remaining west of 90°W, there were some years where rapid transport eastward and out of the GOM occurred. Fluctuations in the dynamics of ocean currents from year to year could result in significant variability in recruitment and abundance of juveniles within developmental foraging areas. Unfortunately, there are few long-term studies examining the abundance of in-water life history stages of Kemp's ridleys in other regions for comparison to recent trends in the NWGOM. Epperly et al. (2007) examined trends in sea turtle catch rates in North Carolina from 1995 to 1997 and 2001 to 2003, but Kemp's ridleys were the least encountered species, and even though more Kemp's ridleys were caught in two of the

latter three years of the study, the CPUE variability was high and yielded no discernable trends. Sea turtle strandings can provide an additional indicator of Kemp's ridley abundance and distribution over a larger geographic scale, if major factors affecting stranding rates, such as shrimping effort, currents, and observer effort, are taken into consideration (Caillouet et al., 1996; TEWG, 2000). Lewison et al. (2003) reviewed sea turtle stranding records in the western GOM from 1986 to 2000 and reported the highest number of Kemp's ridley strandings in 1994, despite more stringent TED regulations, and significantly higher strandings (22%) during 1994–2000 (with mandatory TED implementation) compared with those in 1986–93 (with voluntary TED implementation). They also determined that elevated stranding rates from 1994 to 2000 were evidence of in-water population increases, because shrimping effort and TED compliance had remained relatively constant. Elevated Kemp's ridley strandings have been reported recently in other locations, including a higher number of Kemp's ridleys captured at fishing piers in the Mississippi Sound (Coleman et al., 2013) and over 1200 turtles, the majority of which were juvenile Kemp's ridleys, found cold-stunned in November to December 2014 in Cape Cod Bay (Gorman, 2014).

Even though the overall trends in relative abundance of Kemp's ridleys in nearshore waters of the NWGOM in the present study were inconclusive, the localized changes in CPUE we observed, coupled with information about distribution and habitat use, revealed important site-specific information about juvenile Kemp's ridleys. Radio-tracking studies on juvenile Kemp's ridleys near Sabine and Calcasieu Passes by Renaud and Williams (2005) characterized these young turtles as habitat faithful to jettied tidal passes, staying within 15 km of their release site during summer months. Yet, low frequency of recaptures across all study sites ($\sim 5\text{--}10\%$), especially after 1998, is likely not due to changes in mortality, as discussed above, but suggests ontogenetic shifts to deeper waters or changes in the attractiveness of study areas (i.e., foraging quality) that could cause Kemp's ridleys to disperse in search of better conditions (i.e., prey). In particular, the observed shift in Kemp's ridley abundance from Sabine to Calcasieu Pass after 1997 and the significant increase in Kemp's ridley CPUE at Lavaca-Matagorda Bay since 1996 may be related to changes in habitat quality, foraging success, or other biotic and abiotic

factors. Blue crabs are the preferred prey of Kemp's ridleys in the western GOM (Shaver, 1991; Werner, 1994), and trawl data collected during this study concurrent to entanglement netting operations (to characterize habitat quality) revealed that declines in Kemp's ridley CPUE at Sabine Pass after 1997 coincided with a reduction in blue crab size at this location (Metz, 2004). Furthermore, a blue crab stock assessment conducted by the Gulf States Marine Fisheries Commission (VanderKooy, 2013) indicated general decreasing or steady trends in fishery-independent estimates of abundance for both juvenile and adult blue crab stocks, while commercial landings have declined throughout the last two decades. The western stock has undergone a strong decline in juvenile abundance since the mid-1980s, and a decline in adult abundance from the mid-1980s until the mid-1990s, after which it has remained relatively stable, but with annual fluctuations.

The lack of recaptured Kemp's ridleys during the present study may simply be due to the limited area covered by the nets (~ 400 m/netting day) compared with juvenile Kemp's ridley home range and core areas in this region. Seney and Landry (2011) calculated 50% kernel density estimate core areas for juvenile Kemp's ridleys in Texas/Louisiana that ranged from 192 to 2650 km². Satellite tracking of six Kemp's ridley juveniles (30.9–46.1 cm SCL) from the Mississippi Sound region also revealed extended movements up to 183 km from capture locations and a mean home range of 8787 km² \pm 2294 SD during a 60-d tracking period (Lyn et al., 2012).

While conservation efforts have contributed to this species' overall population growth since the 1980s (Crowder and Heppell, 2011), recent declines in the rate of increase for annual number of nests and hatchlings are a concern and call for a more comprehensive approach to managing Kemp's ridley recovery efforts (Caillouet, 2014). Although funding and logistic limitations created data gaps in our sampling records, the present study generated the longest in-water dataset on benthic juvenile Kemp's ridleys in the NWGOM. Logistical difficulties and expense of in-water sampling should not be a barrier to gathering additional information on Kemp's ridleys in nearshore developmental habitats (NRC, 2010), especially in light of the reproductive potential these individuals represent to the breeding population (Crouse et al., 1987). Kemp's ridleys spend over 99% of their life at sea, and monitoring all life history stages

must be a management priority. An integrated network of more long-term, consistent, in-water surveys comparable to what has been generated for monitoring programs at sea turtle nesting beaches is needed in the GOM.

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