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Loggerhead sea turtle incidentally captured in Ceará, Brazil and later seen nesting in Quintana Roo, Mexico (see page 16). Photo credit: Banco de Imagens, Projeto TAMAR/ICMBIO_CE.

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Interruption of the Kemp's Ridley Population's Pre-2010 Exponential Growth in the Gulf of Mexico and its Aftermath: One Hypothesis

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This commentary proposes one hypothesis to explain the abrupt interruption of pre-2010 exponential growth in the “conventional index” of Kemp’s ridley sea turtle (*Lepidochelys kempii*) population size in the Gulf of Mexico following the end of the 2009 nesting season, and its aftermath (Figs. 1 & 2) (also see data in Burchfield and Peña 2013, which provides maps of Tamaulipas nesting beaches). The conventional index is the combined annual number of nests (*i.e.*, clutches laid) documented during a nesting season, on three segments of nesting beach in Tamaulipas, Mexico; viz., Rancho Nuevo, Playa Dos-Barra del Tordo, and Tepehuajes (Heppell *et al.* 2005). Nest counts are commonly used to monitor changes in sea turtle populations, but many factors affect the relationship between numbers of nests and nesters, and nesters represent very small and varying proportions of the total population in any given year (Seminoff & Shanker 2008; Bjorndal *et al.* 2011). Nevertheless, the annual Kemp’s ridley nest count obviously depends in large part on the annual number of nesters. The nest count is not complete before the nesting season ends and the count is verified, thus a change in annual number of nests between two consecutive years cannot be determined before the nesting season ends and the nest count is verified in the second of the two years. Therefore, whatever caused the drop in number of nests in 2010 could have occurred any time between the ends of nesting seasons in 2009 and 2010. The combined annual number of hatchlings released from the three index beaches is also documented (Fig. 2), and likewise not final until verified after hatching is considered complete for the season.

Natural and anthropogenic mortalities were expected to occur in all Kemp’s ridley life stages following the nesting season in 2009 (Crowder & Heppell 2011; NMFS *et al.* 2011; Gallaway *et al.* 2013; Heppell *In press*), but the only life stages that could have affected the 2010 nest count were adult females, and subadult females that

matured between the ends of nesting seasons in 2009 and 2010. However, natural and anthropogenic mortalities in adult and subadult females were not expected to reach levels high enough to interrupt exponential growth in annual nests (Crowder & Heppell 2011; NMFS *et al.* 2011; Gallaway *et al.* 2013; Heppell *In press*). After the 2010 drop in nests, the substantial increase in nests in 2011 and the slight increase in nests in 2012 (Fig. 1; see also Burchfield & Peña 2013) were encouraging, but probably resulted from population momentum. They suggested that population growth had quickly resumed (Gallaway *et al.* 2013), and provided hope that exponential growth would soon resume. However, the numbers of nests in 2011-2013 were well below those predicted (Fig. 1).

The drop in nests in 2013, and a preliminary datum for nests in 2014 suggesting a further drop, prompted me to begin drafting this commentary in mid-July 2014. The 2014 drop in nests was confirmed by the 12 August 2014 message posted on CTURTLE entitled “Abrupt Setback for the Recovery of the Critically Endangered Kemp’s Ridley Sea Turtle” (www.lists.ufl.edu/cgi-bin/wa?A0=CTURTLE). I adapted the 1966-2013 time series of nests and hatchlings data in Figs. 1 & 2 from Gallaway *et al.* (2013), Benny Gallaway (personal comm., August 2014), and Gladys Porter Zoo’s 2009-2013 annual reports on the Mexico-U.S. Kemp’s ridley population restoration project. Some data points in Figs. 1 & 2 may differ from those published previously, but this does not materially affect my hypothesis.

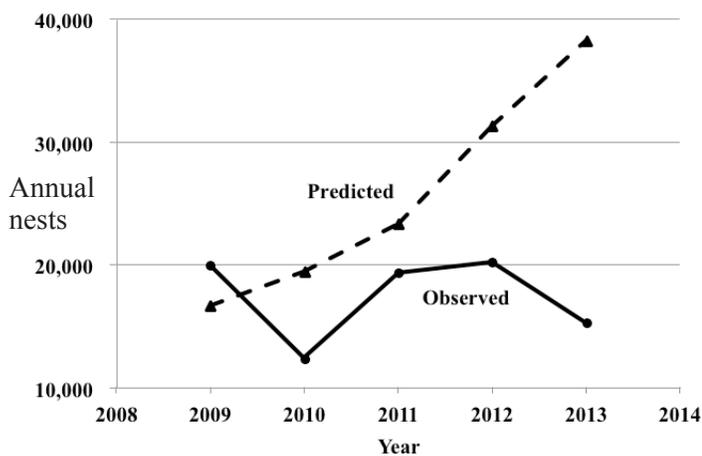


Figure 1. Observed and predicted annual numbers of Kemp’s ridley nests on Rancho Nuevo, Playa Dos-Barra del Tordo, and Tepehuajes beach segments combined, Tamaulipas, Mexico, 2009-2013.

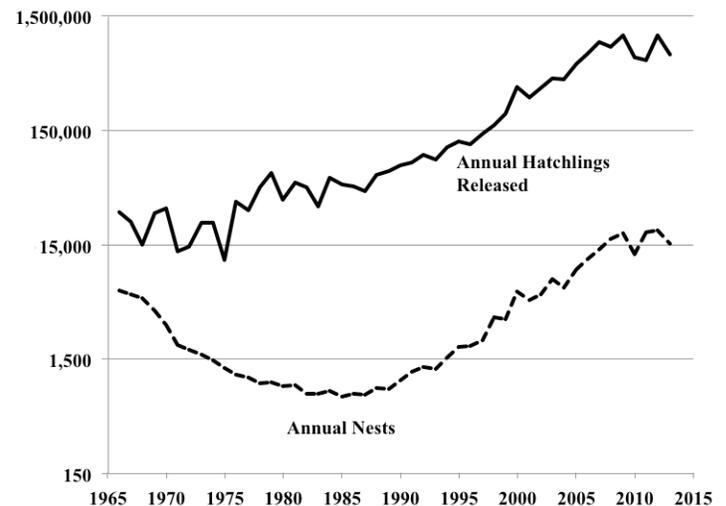


Figure 2. Kemp’s ridley annual nests and annual hatchlings released at Rancho Nuevo, Playa Dos-Barra del Tordo, and Tepehuajes beach segments combined, Tamaulipas, Mexico, 1966-2013. Numbers are in logarithmic scale to show the approximately straight trends during the late 1980s through 2009, which are evidence that the observed increases in numbers within this interval were exponential.

From mid-July through 06 August, I shared my hypothesis and drafts of my commentary with others, and invited some of them to join me as coauthors, but none accepted. I proceeded with the commentary, believing it necessary to focus additional attention on the unexpected crisis facing Kemp's ridley recovery, the urgent need to determine the cause or causes of extraordinary post-2009 changes in population trajectory, and the need for funding to support conservation, monitoring, research, and demographic modeling going forward. The Second International Kemp's Ridley Sea Turtle Symposium (www.kempsridley.info), to be held on 18-19 November 2014, in Brownsville, Texas, "...will provide an opportunity to highlight the status of the species and immediate needs to get it back on track towards recovery."

I hypothesize that the 2009 age distribution and momentum of the Kemp's ridley population in the Gulf of Mexico were fundamentally altered by substantial reductions in numbers of turtles of both sexes at all ages, following the end of the 2009 nesting season (Caillouet 2011). Age distribution is self-explanatory. In addition, population momentum is related to population age distribution. Population momentum was described by Keyfitz (1971) (see also Koons *et al.* 2006) and discussed by Heppell *et al.* (2007) and Caillouet (2010, 2011). Koons *et al.* (2006) is particularly *apropos*, because their study of vertebrate populations concluded that "population momentum will play a critically important role in the population dynamics of long-lived, late maturing organisms that are exposed to large changes in environmental conditions that are caused naturally or via anthropogenic actions." In this regard, the data-rich history and current situation of the Kemp's ridley population provide opportunities for study and modeling of anthropogenic and natural environmental effects on a previously rapidly growing vertebrate population. Although my hypothesis covers both sexes, reduced numbers of males do not limit the number of clutches laid, but reduction in numbers of adult males could affect fertilization of eggs (Coyne & Landry 2007). Therefore, my commentary focuses on females, and the baseline for female Kemp's ridley age distribution and momentum were those existing at the end of the 2009 nesting season.

A critical underlying assumption of my hypothesis is that there were no post-2009 reductions in annual efforts expended toward searching for and documenting nests in Tamaulipas as compared to efforts expended in 2009 (Caillouet 2011; Gallaway *et al.* 2013). If searching and documentation efforts decreased after 2009, my hypothesis has no basis. Likewise, failure of this assumption would also affect validity of post-2009 predictions by demographic models (*e.g.*, Crowder & Heppell 2011; NMFS *et al.* 2011; Gallaway *et al.* 2013; Heppell *In press*).

Fundamental alteration of the 2009 age distribution and momentum of the female portion of the Kemp's ridley population represents (hypothetically) a much greater population setback than previously reported (*e.g.*, Crowder & Heppell 2011; Gallaway *et al.* 2013). If my hypothesis is true, conservation efforts going forward may not be sufficient to prevent further decline for a decade or more, because of the time-lag between release of new cohorts of hatchlings and maturation of surviving females from these cohorts. However, without such efforts the decline could be protracted further. For these reasons, currently unused methods of enhancing hatchling production may be worthy of consideration, discussion and debate (see Mrosovsky & Godfrey 2010). USFWS cuts in funding for

conservation efforts on nesting beaches in Tamaulipas, Mexico pose additional challenges going forward (Plotkin & Bernardo 2014; www.tamucc.edu/news/2014/06/060614%20Hartes%20Heroes.html#.VBmXajF0yos). These cuts have been replaced by funding from other sources through 2014, but funding for years 2015 and beyond is uncertain. NMFS *et al.* (2011) listed lack of funding as a threat that could reverse Kemp's ridley population growth.

Crowder & Heppell (2011) and Gallaway *et al.* (2013) applied models to predict post-2009 changes in annual nest numbers, under hypothetical scenarios involving mortality estimated for various Kemp's ridley age groups in 2010. Modeling by Crowder & Heppell (2011) was based on a time series of annual nests and hatchlings ending with year 2003, and it incorporated a hypothetical upper limit of 12,000 nests protected annually in corrals, with the rest left *in situ*. The stock assessment model applied by Gallaway *et al.* (2013) was based on an updated time series of annual nests and hatchlings, it did not place an upper limit on nests protected annually in corrals, and it incorporated a metric of annual shrimp trawling mortality in the Gulf of Mexico. No previous demographic model of sea turtle populations had incorporated a metric of shrimp trawling mortality, despite the finding by Magnuson *et al.* (1990) that incidental capture in shrimp trawls was the most important source of post-pelagic sea turtle deaths at sea. For year 2010, Gallaway *et al.* (2013) estimated that Kemp's ridley mortality attributable to shrimp trawling was 12.6% of annual total mortality (anthropogenic and natural combined) for ages ≥ 9 years, and 4.4% of annual total mortality for ages ≥ 2 years. In other words, the larger percentage of estimated annual total mortality in 2010 was left unexplained, but was not attributable to shrimp trawling.

Under my hypothesis, reductions in numbers at age for Kemp's ridley females following the 2009 nesting season should not be expected to be proportionate. It is more likely that numbers at age were disproportionately impacted. Impacts of anthropogenic and natural factors on Kemp's ridleys depend on spatio-temporal distributions of the turtles of various ages in relation to spatio-temporal distributions of these factors. A simple example is provided by the 2-yr pelagic stage (ages 0-1 years; Witherington *et al.* 2012; Gallaway *et al.* 2013). The pelagic life stage is not vulnerable to incidental capture in shrimp trawls, but all post-pelagic Kemp's ridleys (ages 2 years and older) are vulnerable to shrimp trawling (Gallaway *et al.* 2013). Spatio-temporal distribution of the pelagic stage is determined by locations and timing of entry of hatchlings into the Gulf of Mexico, and dispersal thereafter by oceanic surface circulation (Ogren 1989; Collard & Ogren 1990; Márquez M. 2001; Putman *et al.* 2010, 2013; Witherington *et al.* 2012). Spatio-temporal distributions of post-pelagic life stages and factors that threaten them have been elucidated by mark-recapture, tracking, in-water sampling, observer programs, bycatch in fisheries, and strandings (*e.g.*, Caillouet *et al.* 1996; McDaniel *et al.* 2000; Morreale *et al.* 2007; NMFS & USFWS 2007; NMFS *et al.* 2011; Seney & Landry 2011; Garrison & Sasso 2012; Lewison *et al.* 2013; Shaver *et al.* 2013).

The post-2009 departure of observed annual numbers of nests from those predicted (Fig. 1) obviously reflected a reduction in nesting by females that were already adults and those that became adults after the 2009 nesting season. Clearly, if numbers of adult and subadult females were substantially reduced by higher than expected mortality, this could explain reduced numbers of nests

in 2010 and subsequent years as compared to model-predicted numbers of nests. However, nesting also could have been reduced by non-lethal factors that prevented migration to nesting beaches, egg production, or both (Benny Gallaway, personal comm., August 2014). Migration from foraging areas to nesting beaches and production of eggs require reserve energy, and in any given year some adult females in the population may have insufficient energy reserves to support migration or production of eggs (Márquez M. 2001; Witzell *et al.* 2005, 2007; Morreale *et al.* 2007; Rostal 2007; Shaver *et al.* 2013; Shaver & Caillouet In press). Energy demands for migration and egg production must be substantial, because Pritchard (1980) observed "...that nesting ridleys invariably look extremely lean, without bulging fat around the limb bases typical of captive animals, and the neck always looks scrawny with the space under the front of the carapace cavernous and shrunken." It is therefore conceivable that post-2009 annual numbers of nests (Fig. 1) reflected failure of large numbers of potential nesters to nest in 2010 and beyond because they became debilitated, undernourished, or experienced reproductive failure for other reasons following the 2009 nesting season (*e.g.*, see www.fws.gov/contaminants/fws_oscp_05/fwscontingencyappendices/L-WildlifePlans/turtle.pdf; Caillouet 2010, 2011; Gallaway *et al.* 2013; Shaver *et al.* 2013; VanderKooy 2013). Examination of available strandings data for subadult and adult female Kemp's ridleys in years 2009-2014 and beyond, including live-strandings and fresh carcasses, could shed light on theory that survivors were unable to nest for whatever reason.

In 2009, prospects for Kemp's ridley recovery were promising. Intensive conservation efforts in Tamaulipas had reversed the previous population decline by 1986, and the nesting range had expanded (Caillouet 2010; Caillouet *et al.* In press; Shaver & Caillouet In press). The use of turtle excluder devices (TEDs) in shrimp trawls and declining shrimping effort had accelerated the population's growth (Heppell *et al.* 2005, 2007; Caillouet 2006, 2010; Gallaway *et al.* 2013). Annual nests and hatchlings had been increasing exponentially for more than 2 decades (Fig. 2). Hatchling inputs had restored the population's age structure, and increased the number of cohorts among females nesting in a given year. By 2009, subadult and adult female life stages represented more cohorts but substantially fewer individuals than all younger life stages combined, which represented fewer cohorts but substantially more individuals (see Seminoff & Shanker 2008). NMFS *et al.* (2011) characterized post-2009 expectations for recovery of the Kemp's ridley population as follows: "We anticipate that the Kemp's ridley will attain its downlisting criterion of 10,000 nesting females in a season by 2011. Based on population growth rates of 19% per year, we anticipate that the Kemp's ridley could attain its delisting criterion of an average of 40,000 nesting females per season over a 6-year period by 2024." USFWS also was optimistic (www.fws.gov/endangered/news/bulletin-summer2009/brighter-future-for-kemp.html): "The Kemp's ridley nesting population is increasing, and we are cautiously optimistic that the species is on its way to recovery."

The cause or causes of departure of observed annual numbers of nests from those predicted (Figs. 1 & 2) remain to be determined. The Deepwater Horizon oil spill and remedial actions taken to mitigate it (Antonio *et al.* 2011; Belter 2014; Fikes *et al.* 2014), incidental capture in shrimp trawls (sero.nmfs.noaa.gov/protected_resources/sea_turtles/documents/shrimp_biological_opinion_2014.pdf), other anthropogenic threats, and environmental phenomena are being considered as possible contributors (Caillouet 2010, 2011; Crowder & Heppell 2011; Fikes *et al.* 2014; Gallaway *et al.* 2013; NMFS *et al.* 2011; Safina 2011).

Also to be considered is a possible effect of Gulf of Mexico carrying capacity for Kemp's ridleys (Gallaway *et al.* 2013). Typically, a population's growth curve is sigmoid in shape (symmetrical or asymmetrical). Once an exponentially growing population passes the inflection point on its growth curve, growth continues but at a diminishing rate as the population approaches an upper asymptote; the magnitude of the asymptote is related to environmental carrying capacity. At a stakeholders meeting (either April 2004 or February 2006) held by the Kemp's Ridley Recovery Team (www.fws.gov/kempstridley/meetingschedule.html) in Houston, Texas, Peter C.H. Pritchard suggested that the inflection point on the Kemp's ridley population growth curve might occur at a high population level, as a consequence of many years of intensive conservation efforts applied toward its recovery (Caillouet 2006), implying that the population might overshoot environmental carrying capacity. A population crash might result from such a situation. Heppell *et al.* (2007) recognized that reductions in habitat and prey availability may have decreased the maximum population size that can be attained. They also recognized that reductions in carrying capacity can result from limitations of available nesting habitat and prey. Furthermore, they noted that population growth could not continue indefinitely because of changes in egg survival rates as a result of decreased *per capita* protection on nesting beaches, possible density-dependent changes in survival and growth, and the potential for increased natural or anthropogenic threats. If carrying capacity for the Kemp's ridley population was abruptly reduced in 2010 by impacts on Kemp's ridley prey species (Caillouet 2010, 2011; Shaver *et al.* 2013; VanderKooy 2013), this could have been a factor influencing post-2009 changes in annual numbers of nests.

The product of annual number of nests and annual average number of hatchlings released per nest estimates reproductive output of nesters in a given year (Witzell *et al.* 2005, 2007; Heppell *et al.* 2007). Annual average number of hatchlings released per nest peaked in 1989, thereafter declining to a little more than half the 1989 level by 2013 (Fig. 3). The decline could represent reduction in

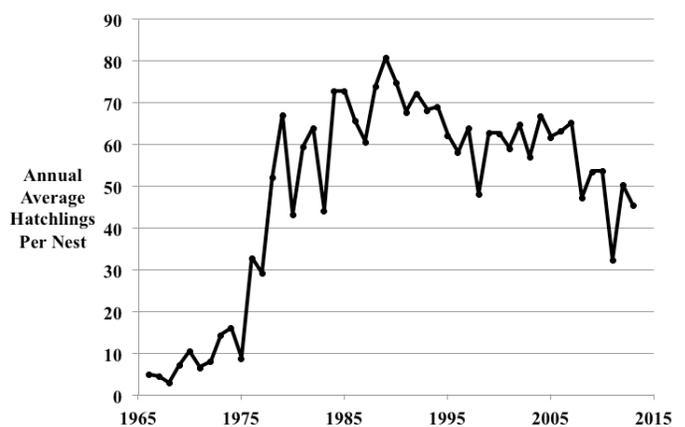


Figure 3. Annual average hatchlings per nest for Kemp's ridley nests on Rancho Nuevo, Playa Dos-Barra del Tordo, and Tepehuajes beach segments combined, Tamaulipas, Mexico, 1966-2013.

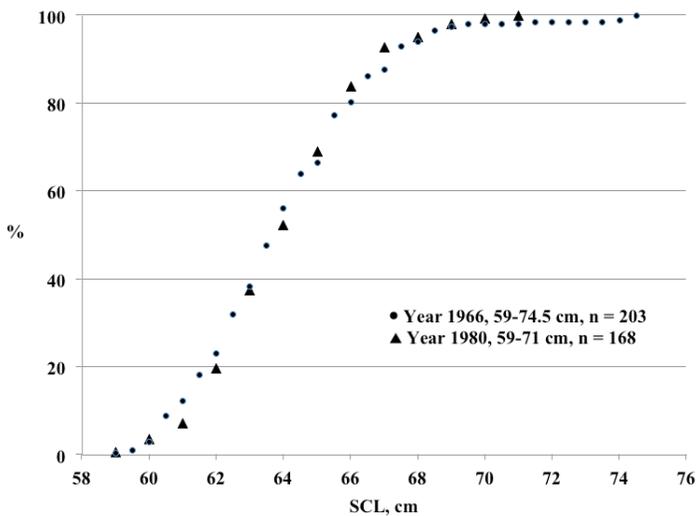


Figure 4. Cumulative distributions (%) of SCL (cm) of Kemp's ridley nesters at Rancho Nuevo, Tamaulipas, Mexico, in 1966 (solid dots) and 1980 (solid triangles). Data adapted from Chavez *et al.* (1968) and Pritchard (1980), respectively.

fecundity of nesters as well as many factors affecting clutch survival rates (Márquez-M. 1994; Witzell *et al.* 2005, 2007; Heppell *et al.* 2007). A long-term trend toward younger, less fecund nesters would be expected as a result of exponentially increasing annual numbers of hatchlings released (Fig. 2). The decline in annual average number of hatchlings released per nest emphasizes the need for time series of annual numbers of first-time (or neophyte) nesters, and their annual proportion of all nesters. Regardless of its cause, the observed decline in annual average number of hatchlings per nest suggests it will now take about twice as many nests to produce a given number of hatchlings as it did in 1989. This will be a further challenge to restoration of exponential population growth going forward.

An obvious question arises as to how my hypothesis can be tested. Sample annual age distributions of Kemp's ridleys would be useful in testing the hypothesis, and they could also be used to validate demographic models. However, annual age distributions are difficult if not impossible to determine by sampling. Direct methods (*e.g.*, skeletochronology, based on dead specimens) and indirect methods (*e.g.*, based on mark-recapture and strandings data) have been used to determine ages of Kemp's ridleys (Snover *et al.* 2007; Avens & Snover 2013; Gallaway *et al.* 2013). Estimated relationships between post-pelagic female Kemp's ridley age and straight carapace length (SCL) are numerous, but they do not apply as well to estimating ages of adults as they do to estimating ages of juveniles and subadults, since growth slows or stops after Kemp's ridleys reach maturity (Snover *et al.* 2007; Caillouet *et al.* 2011; Avens & Snover 2013). Nevertheless, available annual SCL measurements from all sources can be converted to age using available SCL-age relationships (Snover *et al.* 2007; Avens & Snover 2013; Gallaway *et al.* 2013). Statistical comparisons can then be made among annual age distributions derived from each SCL-age relationship applied to SCL measurement data from various sources, or all sources combined. Annual age distributions derived by converting SCL to age can also be compared to model-generated annual age distributions. These approaches can also be applied to nesters only. Model-generated annual age distributions of nesters

can also be used to predict the proportion that first-time nesters represent of total nesters in a given year. Comparisons of model-generated age distributions of nesters among years 2009-2014 and beyond are possible.

Comparing annual SCL distributions of nesters, derived from annual samples taken on Tamaulipas beaches over the years, could also be informative in testing my hypothesis, validating demographic models, and validating SCL-age relationships. For example, demographic models can be used to generate annual SCL distributions, which can be compared to annual SCL distributions obtained from samples from various sources, or all sources combined. The need for recent annual SCL distributions of nesters was recognized by Gallaway *et al.* (2013), and a large sample of SCL measurements of nesters in Tamaulipas was obtained in 2014 (Benny Gallaway, personal comm., August 2014). If nester SCL measurement data are available for other years, they could be valuable for making annual SCL distribution comparisons among years, especially 2009-2014.

I constructed cumulative distributions of SCL for Rancho Nuevo nesters in 1966 and 1980, with data adapted from Chavez *et al.* (1968) and Pritchard (1980); they did not differ very much (Fig. 4), except for the absence of females > 71 cm SCL in the 1980 sample. The 1966 sample was made up of residual nesters remaining when conservation efforts began at Rancho Nuevo, and the 1980 sample contained residual nesters plus younger nesters added as the result of these conservation efforts (see Márquez-M. 1994). Márquez-M. (1994) tabulated maximum, median, and minimum SCL and sample size for years 1966-1992 (Figs. 5 & 6). Median SCL showed little variation, but maximum and minimum SCL varied quite a bit, in part due to wide variation in sample size. The wide range in SCL of nesters representing multiple cohorts in a given year, coupled with limited variation in their median or mean SCL over years (Márquez-M. 1994, 2001), suggests to me that the SCL distribution of neophyte nesters may be indistinguishable from that of nesters representing all accumulated cohorts, with the possible exception that neophytes have not had time to grow to the larger SCLs that are possible with much older nesters. Thus it is important to

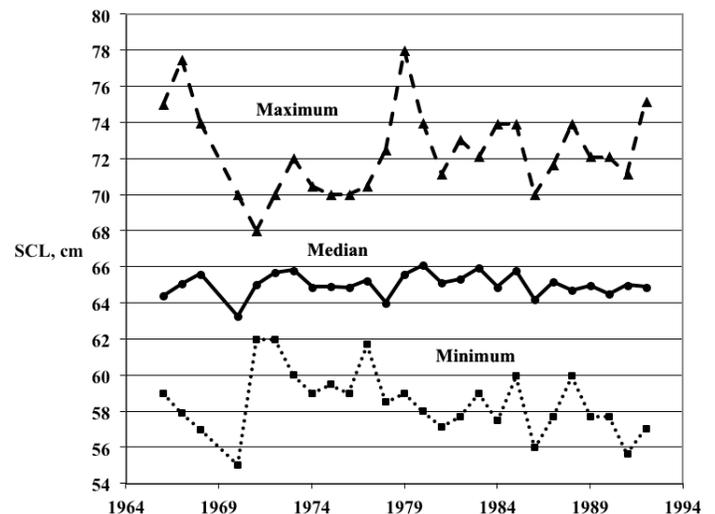


Figure 5. Annual maximum, median, and minimum SCL (cm) of Kemp's ridley nesters at Rancho Nuevo, Tamaulipas, Mexico, during 1966-1992. Adapted from Marquez-M. (1994).

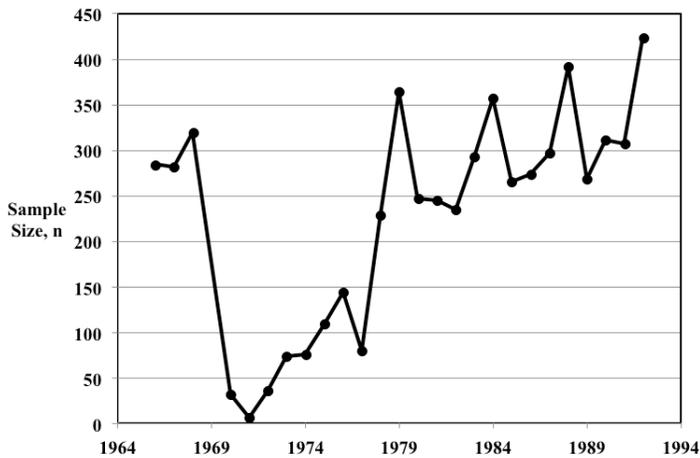


Figure 6. Sample size (n) for SCL (cm) measurements (see Figure 5) of Kemp's ridley nesters at Rancho Nuevo, Tamaulipas, Mexico, during 1966-1992. Adapted from Márquez-M. (1994).

identify neophyte nesters from other nesters in a given year. In fact, Márquez-M. (1994) wrote that "...in the wild, size is not that important in determining sexual maturity." He noted that females nesting at Rancho Nuevo and in captivity can be as small as 55 cm SCL, and concluded that "...age and size at initial maturity are so variable that they should be checked annually in order to conduct meaningful population analysis." Genetic differences among females that survive to maturity, and differences in their histories of exposure to factors that affected their growth, can lead to a wide variation in SCL and age at maturity and first nesting (Márquez-M. 1994). Demographic modelers have not incorporated such variation into their model runs; instead, they have assumed a fixed age at maturity. It is not likely that all surviving females of a given cohort mature simultaneously. Gallaway *et al.* (2013) included an optional maturity schedule or ogive in their stock assessment model, so that variation in age and SCL at maturity could be accommodated in the future. However, this option was never used. In this regard, the approach recommended by Heino *et al.* (2002) to relate age and size at maturity may be worthy of application to Kemp's ridley population modeling.

Examination of SCL distributions of Kemp's ridleys strandings in the Gulf of Mexico will also be useful in testing my hypothesis. For example, the annual SCL distributions of Kemp's ridleys strandings in the Gulf of Mexico in 2010 and 2011 were strikingly different from that in 2009 (Gallaway *et al.* 2013). Going forward, it will be important to examine annual SCL distributions of strandings and to make statistical comparisons among years, especially 2009-2014 and beyond.

Whether or not my hypothesis is true, there can be no doubt that the Kemp's ridley population changed after the 2009 nesting season, and these changes likely will not be quickly rectified. Kemp's ridley recovery has been postponed. Questions remain as to the cause or causes of this setback, what should be done to restore exponential growth in the population, how it will be funded, and how long it will take. I sincerely hope that my hypothesis is wrong, and that sustainable growth in the Kemp's ridley population will soon resume!

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Two More Cases of Green Turtles (*Chelonia mydas*) in the Italian Waters of the Northwestern Adriatic Sea and an Inorganic Contaminant Investigation

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The green turtle (*Chelonia mydas*) is listed as Endangered on the IUCN Red List of Threatened Species (www.iucnredlist.org); they appear to be rare in Italian waters and even more so in the Adriatic Sea, however there is insufficient information due to limited data for the area. Only 28 green turtles were reported in Italian coastal waters from 1986-2008 (Bentivegna *et al.* 2011) and only 12 turtles since 1830 have been reported in the Adriatic Sea (Lazar *et al.* 2004). The Adriatic Sea may play an important role in the foraging and developmental ecology for marine turtles in the Mediterranean due to shallow and warm water (Vallini *et al.* 2011; Casale *et al.* 2012). As such, monitoring the population status and health of green turtles in this region is of great importance.

The Adriatic Sea, due to its peculiar characteristics (small surface area, warm temperature, shallow waters, dense population

of inhabitants, tourists, fisheries and inputs from the Po River) is an important hotspot in term of the impact of humans (Halpern *et al.* 2008; Lazar & Gracan 2011).

Inorganic contaminants (*e.g.*, heavy metals) and persistent organic pollutants (*e.g.*, polychlorinated dibenzo(p)dioxin and furan-PCDD/F-and polychlorinated biphenyls-PCBs) are present in aquatic systems worldwide as a result of their widespread usage and long-range transport by currents (D'Ilio *et al.* 2011). The bioaccumulation within food webs of these toxic substances is a matter of critical concern and the associated impacts on wildlife species in the marine environment must be examined. Marine turtles are long-lived vertebrates that may bioaccumulate contaminants from food, sediment and waters thus making these organisms of increasing interest as potential bioindicators for pollution in



Figure 1. The first specimen of *Chelonia mydas* found in 2009 near Lido delle Nazioni.



Figure 2. The wounds on the carapace of the first specimen of *Chelonia mydas* found in 2009.

marine ecosystems. Persistent organic pollutants (POPs) become concentrated in fatty tissues, are transferred among organisms via food webs, and as such are capable of travelling long distances from the source. POPs exist for many years within the environment and are linked with serious health effects in humans and other living organisms, even at low-level exposure. According to some authors (D'Ilio *et al.* 2011), chromium, together with others metals (*e.g.*, Cu, Fe, Ni) play an essential role in animal metabolism and growth but a chronic exposure to organochlorine contaminants affect biological process in reproductive, endocrine, immune and developmental systems (Keller *et al.* 2004; Keller *et al.* 2006; Lazar *et al.* 2011). Toxic elements such as cadmium and lead could be transferred from a female to her eggs. Cadmium typically undergoes long-term accumulation in the kidneys, principally through binding by metallothionein. Mercury has been recognized to induce toxic effects in fish, including neurotoxicity, impaired growth and development, reduced reproductive success, liver and kidney damage and immunomodulation. For marine turtles, PCDD/F, PCBs and perfluorinated compounds are the most studied POPs but at present only a handful of studies are available.

Here we present results of samples taken from two juvenile green turtles found in the northwest Adriatic Sea adjacent to Italy, one in September 2009 and another in October 2010.

First Case: 22 September 2009. A juvenile green turtle was live-stranded on the coastline of Lido delle Nazioni (Ferrara, Italy; 44°40'N 012°14'E). This individual had a curved carapace length (CCL) of 29.5 cm, straight carapace length (SCL) of 27.0 cm, curved carapace width (CCW) of 28.5 cm, straight carapace width (SCW) of 24.0 cm, and an estimated weight of 4.0 kg (Fig. 1). This turtle showed deep wounds on the 4th costal and 4th and 5th vertebral scutes on its carapace (Fig. 2) and also on the femoral and anal scutes of the plastron (Fig. 3). The animal died after a short time in recovery and a necropsy was carried out with samples of muscle, liver and fat collected in order to analyze heavy metals, dioxins (PCDD/F) and polychlorobiphenyls (PCBs). Some algae and seven barnacles *Chelonibia testudinaria* (about 4 cm in diameter) were found on the carapace, with one barnacle on the prefrontal scales. A necropsy on this specimen revealed that it was a female in good body condition. The distal end of the carapace had wounds of traumatic origin, but it is difficult to determine whether it was due to the impact with a propeller or a boat; this type of impact injury is the secondary



Figure 3. The wounds on the plastron of the first specimen of *Chelonia mydas* found in 2009.

cause of mortality for marine turtles in this area. The distal end of the plastron showed two big holes and injuries with penetration of the bone splinters in the soft tissues of the coelomic cavity. The stomach was full of food consisting mainly of macroalgae (*Ulva lactuca*) and some small shellfish.

The heavy metals analyzed included: lead (Pb), cadmium (Cd), chromium (Cr) and mercury (Hg). The muscle showed quite low concentrations of heavy metals (lower concentration 0.031 mg/kg for lead, upper concentration 0.054 mg/kg for mercury); generally the concentrations of these elements tend to be low in this tissue. It is known that the majority of mercury in the muscle tissues of marine organisms occurs as methyl mercury while in the liver the main form is inorganic (Storelli & Marcotrignano 2003). The liver showed higher concentrations (Pb: 0.133 mg/kg, Cd: 1.421 mg/kg, Cr: 0.016 mg/kg, Hg: 0.264 mg/kg). Unlike Hg, Cd is not biomagnified and is usually found in high concentrations in marine vertebrates with a long life span for example, marine mammals whose primary food source is squid. Squid is an important vector of Cd for predators at the top of the food chain (Jerez *et al.* 2010).

In sea turtles, PCDD/F, PCBs and perfluorinated compounds are the most studied POPs. Unfortunately no threshold level of toxicity for PCBs, PCDD/F exist for sea turtles and each congener exhibits a different toxicity. Some data exist for PCBs but to date only a few studies have been established for PCDD/F levels in green turtles. Analysis of PCDD/F and PCBs were performed on turtle fat: the concentrations of dioxins (17 congeners) were 2.45 pg/g wet weight. On the other hand, the concentrations of PCBs that were dioxin-like (12 congeners) were 12.35 pg/g wet weight. The PCDD/F profile was dominated by penta and hexa chlorinated congeners with a prevalence of furans rather than dioxins. The PCB dioxin-like profile showed the same predominance of penta and hexa chlorinated congeners with a higher concentration of PCB 118.

Second Case: 12 October 2010. A juvenile turtle (Fig. 4) was incidentally caught in a mid-water paired trawl within 0.25 NM of Porto Garibaldi (North Western Adriatic Sea, 44°40'N 012°15'E). This individual had a CCL of 38.5 cm, SCL of 36.5 cm, CCW of 36.0 cm, SCW of 32.2 cm, and weighed 5.5 kg. About 30 barnacles *Chelonibia testudinaria* (approximately 3 cm in diameter) were found on the carapace; 10 were found on the plastron. Following data collection by researchers and just before its release, the turtle was tagged with two A.R.C.H.E. 681 Monel Tags (National Band



Figure 4. The second specimen of *Chelonia mydas* found in 2010 near Porto Garibaldi.

& Tag Co., Newport, KY, USA): IT 3355 in the right front flipper and IT 3357 in the left front flipper. This turtle was released after being tagged from the beach of Porto Garibaldi during the same day of capture and no samples were collected; it has not been seen since then.

The presence of *Chelonia mydas* at these latitudes is extremely rare; the most recent turtle recorded before this study was in 1996. Among the 12 total records found from 1830 in all Adriatic Sea waters between Italy and Croatia, only one of them was found in the Po River Delta, the same area of this study, in 1985 (Lazar *et al.* 2004). The carapace lengths of the two specimens recorded in our study fall in the same range of carapace length previously reported for the Adriatic Sea. All specimens of *Chelonia mydas* that have been found in the Northern Adriatic were juveniles; it has therefore been suggested that Lakonikos Bay in the Ionian Sea is a developmental habitat for this turtle (Margaritoulis & Teneketzis 2003) and the southern Adriatic may contain pelagic habitats for this species (Lazar *et al.* 2004). Green turtles here might use seagrass beds all around the Italian peninsula (Short *et al.* 2007) as neritic foraging habitats (Bentivegna *et al.* 2011) and the anticlockwise current from the Ionian Sea to the North Adriatic. However, this is just an hypothesis because the number of green turtle records is quite low, and also because juvenile *Chelonia mydas* are sometimes still erroneously identified as loggerheads, *Caretta caretta*.

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Sea Turtle Strandings and Mortalities on the Southeast Coast of Guatemala

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Guatemala's Pacific waters provide important migratory corridors and diverse feeding areas for several of the world's species of marine turtles. The olive ridley (*Lepidochelys olivacea*), leatherback (*Dermochelys coriacea*), eastern Pacific green (*Chelonia mydas*) and hawksbill (*Eretmochelys imbricata*) turtles are all known to frequent the coast of Guatemala (Chacon 2000; Higginson 1989). *L. olivacea*, known locally as *parlama*, is the most abundant species of sea turtle in the region and is principally exploited for its eggs, which are sold for human consumption (Brittain *et al.* 2007; Higginson 1989). It is estimated that almost 100% of eggs laid on the 254 km of coastline are harvested and approximately one quarter of these eggs are relocated to local hatcheries under the government's national conservation program (Brittain *et al.* 2007).

Despite the intensity of egg collection, it is generally considered an offense to intentionally kill an adult turtle, as their role in egg production is valued economically. Some commercial fishing enterprises are an exception to this; occasionally offshore shark fishermen use incidentally caught *L. olivacea* to bait their longlines and some commercial fishermen also remove eggs from captured gravid females (Brittain *et al.* 2007; Higginson 1989). In the large fishing port of San Jose, there is also an illegal trade in turtle oil, which is extracted by sun-drying the carcasses of bycaught turtles; the oil is then sold as medicine to relieve respiratory problems (pers. obs.).

"Strandings," the term used to define either dead or live turtles in a weakened condition that wash ashore (Shaver & Teas 1999), occur on a regular basis along Guatemala's Pacific coast. At present, there is no national stranding program or network to reliably document pertinent data in order to identify sources of mortality,

negative interaction between humans and sea turtles, sex ratios, or species distribution. The present situation in Guatemala involves independent, non-governmental organizations (NGOs) collecting and recording non-standardized data on sea turtle mortalities. Generally, only hatchery information (number of eggs buried, incubation period and hatching success) is collated and analyzed at a national level, thus baseline data on Guatemala's sea turtle populations are scant.

Sea turtle strandings data were collected along the Guatemalan Pacific coast during the 2005-2007 and the 2011-2013 nesting seasons (July to December). During the 2005 and 2006 seasons, the Project Parlama initiative, funded under the European Union's European Voluntary Service scheme, conducted standardized stranding surveys at three sites: El Gariton, Hawaii and La Barrona (Fig. 1, Table 1). In the 2007 nesting season, European Union funding ceased and Project Parlama volunteers were only able to survey Hawaii and a new site with assistance of the Guatemalan navy at the navy base near Puerto San Jose. During nesting seasons 2011-2013, UK registered NGO Akazul surveyed La Barrona and in the 2011 and 2012 seasons staff from the Wildlife Rescue and Conservation Association (ARCAS) surveyed Hawaii. Additional strandings records were collected opportunistically at a further six sites outside the routine survey areas in response to reports from the general public. During the May 2011 stranding event, coastal community members and fishermen alerted project staff to the large number of live stranded turtles washed up on the beach or floating at sea.

All data were collected on a standardized data collection form, which required the observer to record information on species,

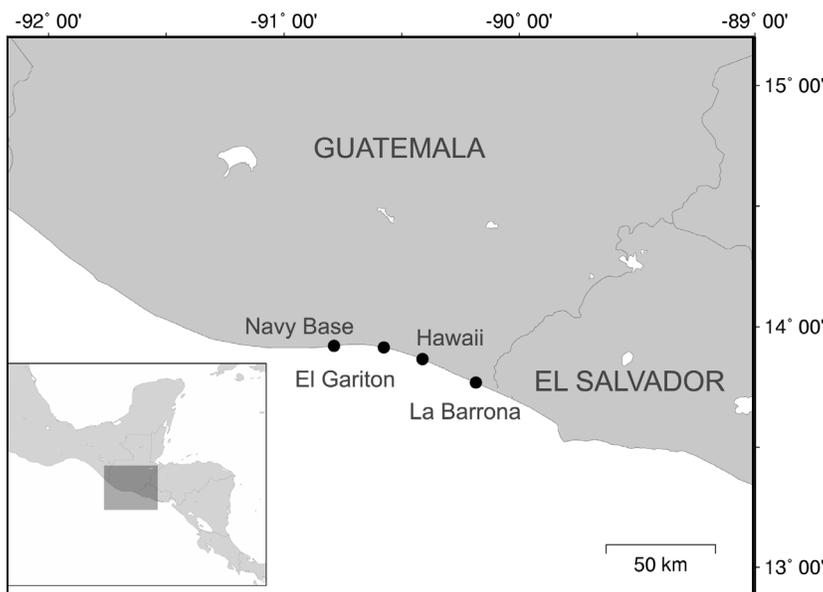


Figure 1. A map of the study sites where sea turtle strandings were monitored and recorded.

Study site	Latitude	Longitude	Km	Yrs monitored
Hawaii	13°51'57"	-90°24'40"	8	2005-2007 2010-2012
El Gariton	13°54'50"	-90°34'33"	8	2005-2007
La Barrona	13°46'06"	-90°11'06"	7.5	2005-2007 2011-2013
Navy Base	13°55'14"	-90°47'15"	1	2007

Table 1. Study sites, including their latitude and longitude, length and years monitored.

Year	Dead turtles	Km monitored	Months monitored
2005	35	30	Jul - Jan
2006	30	30	Jul - Jan
2007	29	9	Jul - Jan
2010	2	7.5	Dec
2011	64	15.5	Jul - Jan
2012	28	15.5	Jul - Jan
2013	67	7.5	Jul - Jan

Table 2. The number of dead turtles recorded by year, site and month of monitoring.

carapace length and width (Curved Carapace Length or CCL and Curved Carapace Width or CCW). We also recorded detailed external observations of the animal, including drawings and photographs to aid written descriptions of any injuries and their location, and the condition of the animal (alive without injuries, alive with injuries, fresh dead, moderately decomposed, severely decomposed, dried carcass, or bones). We recognize there are several ways to categorize turtles into age classes; however, maturity status and sex identification using carapace length and tail elongation have not been studied in *L. olivacea* (Ishihara & Kamezaki 2011). Therefore we used the minimum size (57 cm) of nesting females at La Barrona (Brittain *et al.* 2013) to differentiate between juveniles and possible adults. On relatively fresh carcasses, a tissue sample was collected for use in future regional genetic analyses. Experienced volunteers performed necropsies and as available resources and facilities were limited, only very basic information was obtained. For example, any visual abnormalities or foreign bodies, such as hooks, line, or plastic discovered in the gastrointestinal tract were recorded.

A total of 256 turtles (34 alive and 222 dead) were recorded as stranded along the 28 km study area of Guatemala's Pacific coast during 2005-2007, 2010 and 2011-2013, respectively (Table 2). The most frequently stranded turtle was *L. olivacea* (n = 229), followed by *C. mydas* (n = 20) and *E. imbricata* (n = 5). For two turtles, the species could not be determined due to decomposition of the carcasses. Peak months of *L. olivacea* strandings occurred between June and August, coinciding with the beginning of nesting season; the majority of strandings were female turtles (Fig. 2).

Curved carapace length (CCL) measurements for 140 turtles were obtained (54.7% of the total), allowing us to estimate the state of maturity inferred by the carapace length of the individuals stranded. Of the 116 olive ridley turtles measured, the mean CCL was 63.6 ± 4.6 SD cm (range: 45-75 cm) (Fig. 3). The minimum carapace length of females nesting at La Barrona, Guatemala is 57 cm (Brittain *et al.* 2013) suggesting that 94.9% of animals captured were of reproductive adult size. Of *L. olivacea* strandings 48% were female, 27% male and the sex was undetermined in 24% of strandings mainly due to decomposition and scavenging of the specimens (Fig. 2). The mean CCL of hawksbill turtles stranded was 36.7 ± 2.0 SD cm (range: 35-39.5 cm; n = 5) and based on minimum CCL of nesting females in El Salvador (63 cm) (Liles *et al.* 2011), 100% were categorized as juveniles. For eastern Pacific green turtles measured, the mean CCL was 71.6 ± 14.4 SD cm (range: 35-98 cm; n = 17). Based on minimum size of females nesting at Galapagos with CCL 60.7 cm (Zarate *et al.* 2003), 82.4% of the individuals recorded were possible adults.

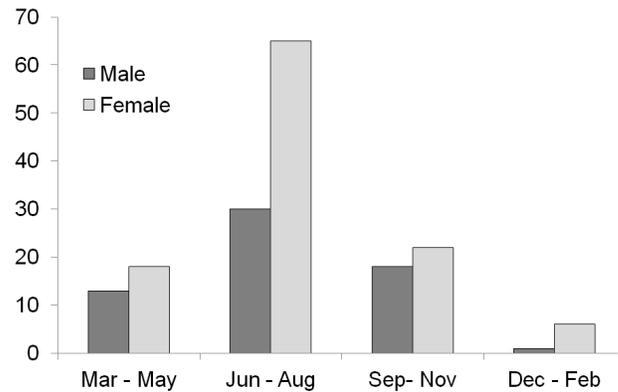


Figure 2. The temporal distribution of stranded *L. olivacea* turtles by sex (n = 173) on Guatemala's Pacific coast during nesting seasons 2005-2007 and 2011-2013.

Two percent of strandings were classified as alive without any visible injuries, 12% alive with injuries, 27% fresh dead, 30% moderately decomposed, 8% severely decomposed, 3% dried carcass, 4% skeleton (bones only) and for 14% of turtles the observer(s) did not record any information. All five strandings recorded as alive without injuries were juvenile hawksbill turtles and were successfully released alive.

The cause of death could not be determined in a majority of the turtles examined due to several factors: severe decomposition or scavenging of specimens by dogs or vultures, inexperienced personnel unable to carry out necropsies, limited resources and no access to diagnostic tests. Four individual turtles displayed evidence of entanglement in fishing gear and a further four were recorded with propeller injuries. Three specimens were found with their plastron and pectoral muscles removed, which is characteristic of injuries inflicted by shark fishers using incidentally caught ridleys as bait for longlines and for turtle oil. Eleven turtles were found with small incisions along the edge of the plastron, which indicates that eggs were removed; this practice has long been considered as evidence of bycatch on commercial fishing vessels (Higginson 1989).

In May 2011 a large number of turtles (n = 34) consisting of 33 *L. olivacea* and one *C. mydas*, stranded over a 20-day period at the study site in Hawaii. Eighteen were alive but of poor body condition and 16 stranded dead. Live turtles carried heavy epibiota loads and were emaciated, which suggested they had undergone prolonged periods of inactivity without feeding. Necropsies were carried out on 16 of the turtles and histological samples were collected for 7 individuals in collaboration with the National Wildlife Health Centre, Honolulu Field station, Hawaii, USA but results were inconclusive (Handy *et al.* In press).

In 2013 between July and December, 67 dead turtles were recorded within 7.5 km of beach at La Barrona. Of these 55 were *L. olivacea* (12 male, 36 female and 7 unrecorded), 11 *C. mydas* (1 male, 6 female and 4 unrecorded) and for one animal the species was unrecorded due to decomposition of the carcass. The cause of death was undetermined in all strandings. Between September and October 2013, 201 dead turtles were observed on the coast of El Salvador (mostly around the La Libertad area). High levels of saxitoxins were present in the tissues of dead turtles and high concentrations of *Gymnodinium catenatum*, a species of phytoplankton that is known to cause harmful algal blooms (HABs), were found in coastal water

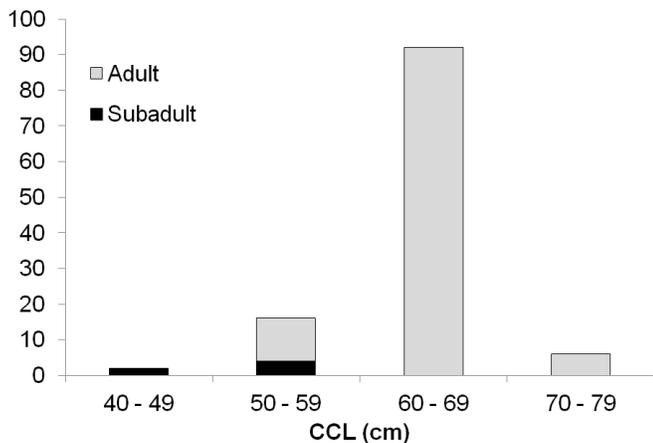


Figure 3. The curved carapace length (CCL) distribution of stranded *L. olivacea* sea turtles (n =116) showing the composition of observed size classes: subadults (45 to 56cm) and apparent adults (>57 cm).

samples (Amaya *et al.* 2014). Previously red tide has caused mass mortality of sea turtles in El Salvador (Barraza 2009) and given the close proximity of La Barrona to El Salvador the high number of turtle deaths in 2013 could be linked to that cause.

Findings from the strandings program show higher numbers of turtle mortalities from June to August. This period coincides with both the beginning of the olive ridley nesting season and encompasses a large part of the shrimp trawling season. It has been documented that bycatch rates of sea turtles are high when there is an overlap between fishing areas and important habitat (Mancini *et al.* 2011; Peckham *et al.* 2007). In Guatemala, adult olive ridleys appear to be vulnerable to commercial fishing activity during nesting season, when they migrate to coastal waters (Morreale *et al.* 2007).

In 1996 a legal requirement was made for all shrimp trawlers to carry turtle excluder devices (TEDs) (CALAS 2006) to reduce the negative impact on sea turtles. At present there exists a general consensus amongst coastal residents and NGOs that sea turtle mortalities in Guatemala are primarily caused by commercial shrimp trawlers that are not using their TEDs. However, there is no biological or physical evidence to substantiate these accusations. Additionally, recent studies in the eastern Pacific have identified small-scale fisheries as cause for concern for impacting sea turtle populations in the eastern Pacific (Alfaro-Shigueto *et al.* 2011). In Guatemala the small-scale commercial fishing sector represents 86% of the overall fleet with over 200 registered vessels targeting shark, dorado and snapper with gillnets and longlines working up to 200 nautical miles offshore (ATP 2004). Koch *et al.* (2013) suggested that the artisanal shark fishery in Baja California Sur working 20 – 40 nm offshore with longlines and gillnets may have large impacts on turtle and marine mammal populations, although few if any of the carcasses generated by this fishery are likely to strand due to currents and beach orientation. Initial communications with Guatemalan fishers working this type of gear in the largest fishing port at Puerto San José would suggest that a significant number of turtles are incidentally caught in the fishery but these mortalities may not be evident in total mortality counts yielded from beach strandings counts.

There is limited information available on temporal or spatial sea turtle distribution in Guatemala's Pacific waters however current

work suggests that olive ridleys are the most abundant species and in some areas may be present year round in up to 15 nm offshore (Brittain, unpublished data). In light of recent studies about small-scale fishery impacts (Alfaro-Shigueto *et al.* 2011; Mancini *et al.* 2011; Peckham *et al.* 2007), further work needs to be carried out to determine the extent and impact of small-scale commercial fisheries on sea turtles in Guatemala as mortalities occurring offshore may not be accounted for in total mortality counts of stranded turtles. It would also be beneficial to carry out regular strandings counts outside of the turtle nesting season to further understand temporal distribution patterns of strandings and to determine overlap with commercial fishing activity. This information has great potential to improve knowledge of sources of sea turtle mortality in Guatemala and highlight areas where stronger conservation measures are required.

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Flying Fish Egg Harvest off Keelung, Taiwan Uncovers Occurrence of Pelagic-Phase Green Turtles

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Fish eggs are considered a nutritionally rich delicacy by humans worldwide and especially in East Asia, including Taiwan. Flying fish eggs that are flavored and colored are known as “tobiko.” Sea turtles are also known to seek out and consume fish eggs in oceanic habitats (Fritts 1981; Richardson & McGillivray 2001; Parker et al. 2005, 2011; Witherington et al. 2012). However, the extent and details of such foraging, like many aspects of sea turtle surface-pelagic ecology, have not been widely documented. In March 2014, we had the unique opportunity to learn about traditional flying fish egg harvest and associated sea turtle foraging habitats from the local Taiwanese fishermen in Keelung (25.131°N, 121.737°E), which is located at the northern end of Taiwan (Fig. 1).

A seasonal harvest for the eggs of flying fish from the family *Exocoetidae* occurs in the offshore waters of Keelung in the Pacific around the time of the Chinese lunar Tuen Ng festival (in Chinese, 端午節), which begins in mid-May and continues to the end of July until the allowable amount of eggs are collected throughout Taiwan is reached (e.g., 350 tons in 2014). The harvest by local fishermen coincides with the occurrence of *Sargassum* drifts, the natural substrate used by flying fish to deposit eggs in masses that hang down into the water column. The fishermen construct artificial rectangular mats or rafts made of rice straw with synthetic foam strips attached along the edges for floatation (Fig. 2A-C). Four strips of rice straw mats 15 cm wide are attached longitudinally along the midline of the rafts and hang down into the water, thereby creating more surface area for egg attachment. These mats, measuring about 2 m by 3 m by 1-2 cm thick, are laid out on the sea surface attached

to one another in areas where floating *Sargassum* mats occur. The location of these artificial mats varies annually and may be up to 50 km offshore, depending on the location of the *Sargassum* drifts. The mats are lifted daily to collect the voluminous quantity of eggs that accumulate on them (Fig. 2D). These mats are used in one season only and replaced with new ones the following year. We learned from the fishermen that during the egg harvest, green turtles (*Chelonia mydas*) of approximate carapace length 20 cm have been observed under the mats, presumably feeding on the fish eggs or other organisms associated with the fishing mats; they may possibly be using the mats as shelter as well. The local fishermen appear not to be bothered by the presence of the green turtles, but instead are more focused on harvesting the fish eggs for their business. We observed that the remains of flying fish eggs were gathered on artificial cotton-like materials, likely ocean debris, during our visit, suggesting that the spawning season of flying fish in this region started as early as March prior to the fishery opening. Such spawning grounds established in floating *Sargassum* or other artificial mats may provide energy-rich food that may be consumed by green turtles.

In the western North Pacific, *Sargassum* drifts that serve as habitats for diversified fish communities are known to occur off the southeast coast of Japan near the Kuroshio Current (Uehara et al. 2006) and in the Tohoku area of northeastern Japan along the Kuroshio Extension (Safran & Omori 1990). Komatsu et al. (2008) also reported that drifting seaweed mats, which were exclusively *Sargassum horneri*, proliferated in March through May and were found in waters off Zhejiang, China between the continental shelf peripheral area and the Kuroshio oceanic front within the eastern East China Sea (Fig. 1). This implies that the seasonal occurrence of *Sargassum* mats in the offshore waters of Keelung that serve as foraging grounds for green turtles may also be affected by the Kuroshio Current.

We highlight the importance of incorporating local people and their knowledge in creating novel opportunities for sea turtle exploratory studies. Such trustful and respectful communication brings insight into what may be the first-documented habitat use of surface-pelagic green turtles in the *Sargassum* drifts and man-made floating mats off Keelung. We intend to visit Keelung again in the future to talk with more fishermen for greater insights into their rich oceanic work life that is shared with sea turtles. In addition, visual surveys by observers on fishing boats should be conducted as far as possible to collect, verify and establish baseline information about sea turtles found in and around the fishing mats. The analyses of genetic composition and oceanographic features may also add insights into the geographic natal origin of these pelagic-phase green turtles.

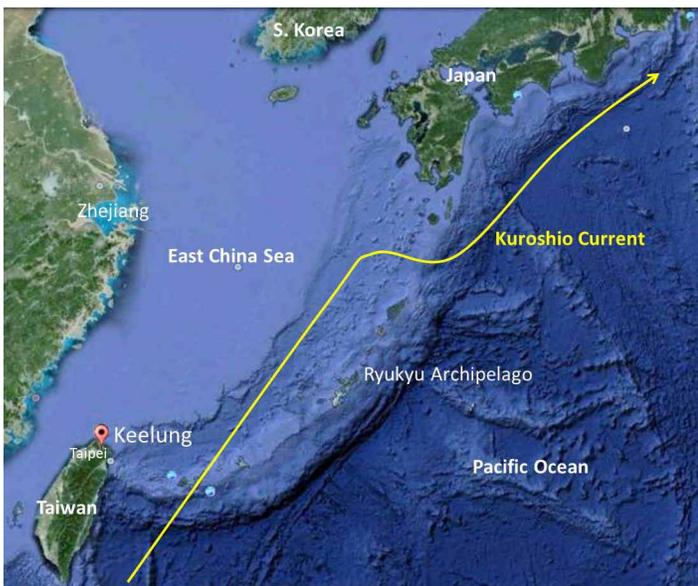


Figure 1. A map showing the locations of Keelung, the East China Sea and the extent of the Kuroshio Current.



Figure 2. Panels A-C: Artificial rectangular mats or rafts made of rice straw with synthetic foam strips used to attract flying fish to deposit eggs. Panel D: Flying fish eggs (roe).

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Second Record of Tagged Loggerhead Moving Between South and North Atlantic

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On 24 May 2013, an adult female loggerhead was observed nesting on X'cabel Beach, in Quintana Roo, Mexico (20.341126 °N, -87.343965 °W). The turtle measured 92.6 cm curved carapace length (CCL) and 81.6 cm curved carapace width (CCW). This turtle had one Inconel tag (type 681, National Band Co., Kentucky, USA) on its left front flipper with the unique identifier BR38046. There were no other tags or tag scars visible. The turtle laid 59 eggs, which were relocated to a central hatchery for incubation (one egg was deformed). At the end of incubation, 21 eggs produced live hatchlings that were released in the ocean; the other eggs remained unhatched.

The turtle had been originally tagged on both front flippers on 08 March 2005 in Almofala, Ceará, Brazil (-2.9010 °S, -39.8378 °W, see cover photo). It had been captured in a fishing weir known locally as a *curral de pesca*. Turtles are regularly captured incidentally in this gear and are released unharmed by local fishers (Marcovaldi et al. 2001). These captures also provide an excellent opportunity for various types of research, including mark recapture, telemetry, and conservation genetics (Marcovaldi et al. 2001; Godley et al. 2003; Naro-Maciel et al. 2007). The carapace size of this turtle increased 15.8 cm over the 8 year period between tagging in Brazil and recapture in Mexico. This corresponds to 1.98 cm/yr growth rate, which is similar to growth rates reported for subadult loggerheads in the NW Atlantic (Braun-McNeill et al. 2008).

Satellite telemetry data have shown that post-nesting loggerhead females from Bahia, Brazil use the coastal waters off of Ceará as foraging areas (Marcovaldi et al. 2010). This is only the second reported tag return of a loggerhead turtle moving across the Equator in the Atlantic Ocean. The other was a captive-raised turtle that was released as a juvenile from Espírito Santo, Brazil and was recaptured 3 years later in the Azores (Bolten et al. 1990). Interestingly, genetic analyses of tissue samples collected from loggerheads foraging in waters of the Rio Grande Elevation, a submarine ridge off of southern Brazil, suggested that some turtles had originated from rookeries in the NW Atlantic and Mediterranean (Reis et al. 2010). It may be the case that as more juvenile loggerheads are tagged in Brazil, there may be more observed tag returns in the future, showing turtles moving from South Atlantic foraging grounds to North Atlantic nesting sites.

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REPORT

China/USA Sea Turtle Workshop in Hawaii 2014

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An invitational China/USA Sea Turtle Workshop was convened in Honolulu August 25-29, 2014 under the auspices of the Bilateral Living Marine Resources (LMR) initiative of NMFS Headquarters and the Chinese Academy of Fishery Science in Beijing. The official hosts and organizers for the workshop were George Balazs (PIFSC), Jeffery Seminoff (SWFSC) and Thierry Work (USGS) assisted by PIFSC John Wang, Shandell Brunson, and JIMAR Denise Parker. The workshop had been formally in the development and planning stages since April 2012, as the result of a LMR-sponsored sea turtle meeting in Shanghai attended by Balazs, Seminoff, and Wang. Nine Chinese region scientists participated in the workshop using non-USA travel funds. Areas represented included Hainan Island, Hong Kong, Taiwan, and Mainland China. Four of the participants were university professors, three were government biologists, one was a graduate student, and one represented an NGO involved in research.

Workshop activities over the five days consisted of mixtures of seminar presentations, conversation exchanging information to build bridges and formulate ideas for future actions, including collaborative research, and actual hands-on field work with Hawaiian sea turtles. Two of the several 'ways forward' agreed upon for cooperation consisted of a follow-up workshop in mainland China in 2015, and reciprocal China/USA visiting scientists for 1-2 month periods.

The workshop was deemed highly successful by all involved. The Chinese scientists were harmoniously engaged under the LMR initiative. Cross-cultural goodwill and cooperation were advanced between China and the USA, using sea turtle science and ecology as the means of person-to-person diplomacy.



Figure 1. Group photo taken on 25 August of participants of the China/USA Sea Turtle Workshop in Hawaii 2014. Left to Right Back Row: M. Murphy, W-C Huang, J. Seminoff, T-H Li, Y. Wang, R. Lo, S. Brunson, J. Wang, W. Liu. Left to Right Front Row: D. Parker, G. Balazs, F. Yeh, H-X Gu, L. Fu, Z-R Xia, T. Work.

ANNOUNCEMENT

2nd Southeast Regional Sea Turtle Meeting, 4-6 February 2015 in Jekyll Island, GA, USA

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I am pleased to announce the second Southeast Regional Sea Turtle Meeting (SERSTM) will once again be held on Jekyll Island, GA on February 4-6, 2015. The meeting, hosted by the Georgia Sea Turtle Center, will include oral and poster presentations on a variety of topics pertaining to sea turtle biology and conservation throughout the southeastern region of the United States. In addition, there will be a whole day dedicated to workshops and a special Sea Turtle Health

and Rehab Workshop conducted by Terry Norton from the Georgia Sea Turtle Center. Everyone who works on sea turtles or sea turtle issues in the region is encouraged to attend. For further information about hotels and to register for SERSTM, please visit the network's website: www.serstm.org. Please follow the network on Facebook at www.facebook.com/SoutheastRegionalSeaTurtleMeeting. I look forward to seeing everyone at the meeting in February.

RECENT PUBLICATIONS

This section is compiled by the Archie Carr Center for Sea Turtle Research (ACCSTR), University of Florida. The ACCSTR maintains the Sea Turtle On-line Bibliography: (<http://st.cits.fcla.edu/st.jsp>). It is requested that a copy of all publications (including technical reports and non-refereed journal articles) be sent to both:

The ACCSTR for inclusion in both the on-line bibliography and the MTN. Address: Archie Carr Center for Sea Turtle Research, University of Florida, PO Box 118525, Gainesville, FL 32611, USA.

The Editors of the Marine Turtle Newsletter to facilitate the transmission of information to colleagues submitting articles who may not have access to on-line literature reviewing services.

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