

# Marine Turtle Newsletter

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An adult loggerhead turtle captured while foraging was tracked during an exceptionally long migration route prior to nesting in Greece. See pages 10-11. Photo by ARCHELON.

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# Excessive Annual Numbers of Neritic Immature Kemp's Ridleys May Prevent Population Recovery

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Nestings of adult female Kemp's ridley sea turtles (*Lepidochelys kempii*) on western Gulf of Mexico (GoM) beaches of Tamaulipas, Mexico, and hatchlings (both sexes) that reached the GoM from these beaches, have dominated this endangered species' total annual reproductive effort and output, respectively (Heppell *et al.* 2005, 2007; Márquez-M. *et al.* 2005, 2018; National Marine Fisheries Service [NMFS] *et al.* 2011; Wibbels & Bevan 2016). In other words, nesters on Tamaulipas beaches have been the dominant source of Kemp's ridleys of all life stages. The primary Kemp's ridley nesting beach near Rancho Nuevo, Tamaulipas (see map Figure 1 in Márquez *et al.* 1999) was discovered in 1947 by Andrés Herrera, who filmed the species' largest ever recorded arribada (Carr 1963; Hildebrand 1963; Pritchard 2007; Bevan *et al.* 2016; Wibbels & Bevan 2016). Hildebrand (1963) estimated there were 40,000 adult females in this arribada, and noted (according to Herrera's observations) that many eggs already laid were dug up by later nesters, thereby saturating the entire nesting zone with eggs easily available to predators such as coyotes (*Canis latrans*). Arribada nesting apparently overwhelms natural predators with an ephemeral overabundance of eggs, leaving the rest to incubate and hatch in comparative safety (Pritchard 2007; NMFS & [US Fish and Wildlife Service [USFWS] 2015). Hatchlings are vulnerable to parasites and predation while in the nest, then to predation during their crawl to the surf (Marquez-M. 1994; Bevan *et al.* 2014). Mortality of hatchlings due to predation by numerous species of marine fish is greater than that from on-beach predation (Carr 1967; NMFS *et al.* 2011; NMFS & USFWS 2015).

Herrera's movie also showed men exploiting eggs (Hildebrand 1963; Carr 1967). However, seven decades before the 1947 arribada, Prieto (1873) reported that marine turtles and their eggs contributed to the commerce of Tamaulipas. In the early 1920s, Mexico's federal government began promulgating laws, regulations, and acts aimed at reducing harvest of sea turtles and their eggs on land and at sea (Marquez-M. 1994; Márquez *et al.* 1998; Márquez-M. *et al.* 2018). Despite such measures, the Kemp's ridley population declined substantially and was still declining when Hildebrand (1963) urged promulgation of conservation measures to prevent extinction of this species.

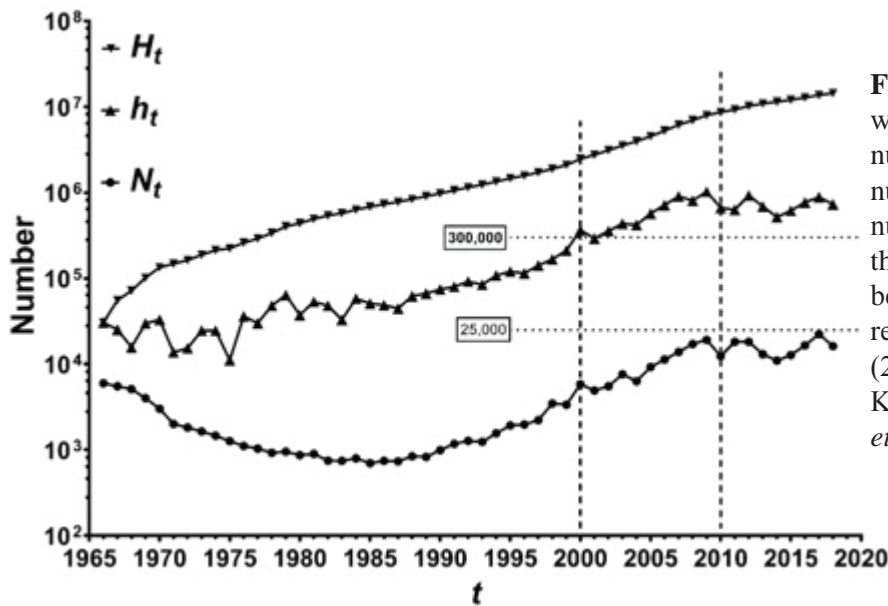
In 1966, Mexico's federal government initiated on-beach patrols and annual protection of as many nesting females, nests, and hatchlings as possible near Rancho Nuevo (Chavez *et al.* 1968; Heppell *et al.* 2005, 2007; Márquez-M. *et al.* 2005, 2018; Pritchard 2007; Márquez-Millán *et al.* 2014). At the same time, Mexico's federal government initiated (1) an annual count of nests ( $N_t$ , where  $t$  is calendar year), which provided an index of annual abundance of nesting females, and (2) a corresponding annual count of hatchlings ( $h_t$ ) released into the GoM, which provided an index of annual reproductive output of these nesting females (Caillouet *et al.* 2015b, 2016, 2018; Wibbels & Bevan 2016). Annual production of hatchlings ( $h_t$ ) was being restored, but annual nests ( $N_t$ ) continued to

decline (Fig. 1), because not enough time had elapsed for the new recruits to reach maturity (Marquez-M. 1994). Carr (1977) called for action to save Kemp's ridley from extinction, noting that the preceding decline in the population was caused by overexploitation of eggs combined with heavy natural predation pressures, but the decline in progress was brought about by incidental capture in shrimp trawls.

In 1978, the US-Mexico Kemp's ridley restoration and enhancement program was initiated (Márquez Millan *et al.* 1989; Márquez-M. *et al.* 2005, 2018; Pritchard 2007; Márquez-Millán *et al.* 2014; Caillouet *et al.* 2015b). The population continued declining to near extinction by 1985 (Fig. 1; Byles 1993). During the 1947-1985 population decline, magnitudes of ecological roles in aquatic and terrestrial habitats (Bjorndal & Bolten 2003; see review by Lovich *et al.* 2018) fulfilled by Kemp's ridley no doubt had diminished substantially, and the GoM ecosystem likely adjusted to declining abundance of all life stages.

In any analysis or modeling of trends in  $N_t$  and  $h_t$  (Fig. 1), consideration should be given to the intermittent increases in length of the Tamaulipas nester-abundance-index beach over the years, from that of Rancho Nuevo exclusively, to the maximum comprising Rancho Nuevo, Tepehuajes and Playa Dos combined (see map Figure 1 in Márquez *et al.* 1999; Turtle Expert Working Group [TEWG] 1998, 2000; Heppell *et al.* 2005, 2007; Márquez-M. *et al.* 2005; NMFS *et al.* 2011; Márquez-Millán *et al.* 2014). Also, during 1966-1977, most nests found and counted ( $N_t$ ) were left *in situ*, thus clutches of eggs that were translocated to protective, on-beach hatcheries represented small proportions of  $N_t$ ; the counts of hatchlings released ( $h_t$ ) in those years originated only from clutches that were translocated and protected (TEWG 1998, 2000; Márquez *et al.* 1999; Márquez M. 2001; Márquez-M. *et al.* 2005). During 1978-2018, eggs from all nests found were translocated to protective, on-beach hatcheries, except for those deliberately left *in situ* (TEWG 1998, 2000; NMFS *et al.* 2011; Gallaway *et al.* 2013, 2016a; Caillouet *et al.* 2016), because either they exceeded the capacity of on-beach hatcheries, or it became logistically impossible to translocate all eggs (Bevan *et al.* 2014). A robust examination of archived records could be helpful in evaluating levels and efficacy of monitoring  $N_t$  and  $h_t$  over the years.

Cumulative beneficial effects of conservation interventions that reduced mortality on Tamaulipas beaches and at sea, combined with other factors, reversed the decline in  $N_t$  by 1986 (Fig. 1; Byles 1993; Marquez-M. 1994; Caillouet 2010; Caillouet *et al.* 2016), and led to rapid increase in  $N_t$  to 19,361 by 2009 (Crowder & Heppell 2011; NMFS *et al.* 2011; Gallaway *et al.* 2013, 2016a, 2016b; Caillouet 2014; Caillouet *et al.* 2015b, 2016, 2018; Dixon & Heppell 2015; NMFS & USFWS 2015; Mazaris *et al.* 2017; Kocmoud *et al.* 2019). The other factors were those that contributed to reduction in mortality associated with shrimp trawling in GoM waters, including the 1976 US-Mexico treaty that phased out US



**Figure 1.** Trends in Kemp's ridley  $H_t$ ,  $h_t$ , and  $N_t$ , where  $t$  is calendar year,  $H_t$  is cumulative annual number of hatchlings released,  $h_t$  is annual number of hatchlings released, and  $N_t$  is annual number of nests (*i.e.*, clutches of eggs laid) on the Tamaulipas, Mexico nester-abundance-index beach during 1966-2018. Horizontal dotted lines represent annual minima,  $h_t$  (300,000) and  $N_t$  (25,000  $\approx$  10,000 adult females), for downlisting Kemp's ridley to threatened status (see NMFS *et al.* 2011).

shrimp trawling in Mexico's GoM waters by 1980, the seasonal Texas Closure to shrimping that began in 1981, the use of turtle excluder devices (TEDs) first required by US federal regulations initiated in 1987 and expanded thereafter, hurricane damage to GoM shrimp trawlers and processing facilities, and deteriorating economic conditions within the GoM shrimp industry (Condrey & Fuller 1992; Iversen *et al.* 1993; Lewison *et al.* 2003, 2013; Caillouet *et al.* 2008, 2016; Nance *et al.* 2008, 2010; Gallaway *et al.* 2013, 2016a, 2016b). Kemp's ridley mortality in all life stages, except the oceanic stage, had been greatly reduced (Márquez Millán *et al.* 1989; Kemp's Ridley Recovery Team 1992; Lewison *et al.* 2003, 2013; Heppell *et al.* 2005, 2007; Márquez-M. *et al.* 2005, 2018; Crowder & Heppell 2011; Finkbeiner *et al.* 2011; NMFS *et al.* 2011; Márquez-Millán *et al.* 2014; NMFS & USFWS 2015; Caillouet *et al.* 2016; Valdivia & Suckling 2019). Post-1985 increases in  $N_t$  and  $h_t$  through 2009 suggested that all Kemp's ridley life stages had increased in abundance, and that their ecological roles and contributions to biodiversity and population resilience within GoM ecosystem were being restored.

The US-Mexico recovery plan (NMFS *et al.* 2011) established the following demographic criteria for downlisting Kemp's ridley status according to the US Endangered Species Act from endangered to threatened status: at least 10,000 females ( $\approx$  25,000 nests) nesting in a season on the nester-abundance-index beach, and at least 300,000 hatchlings released annually from that beach. Horizontal dotted lines in Fig. 1 depict these downlisting thresholds for  $h_t$  and  $N_t$ , and vertical dashed lines connect the points for  $h_t$  and  $N_t$  in 2000 and 2010. The recovery plan's population model predicted that these thresholds would be reached by 2011, and that  $N_t$  would continue increasing at a rate 19% per year through 2020, assuming that survival rates within each life stage remained constant (NMFS *et al.* 2011). However, this rapid increase in  $N_t$  was interrupted in 2010, the year in which the Deepwater Horizon (DWH) oil spill occurred in the northern GoM (Bjorndal *et al.* 2011; Caillouet 2011; Crowder & Heppell 2011; Gallaway *et al.* 2013). Kemp's ridley strandings increased in the northern GoM during 2010 and 2011, and the DWH oil spill and shrimp trawling received the most attention as possible causes (Caillouet 2011; Gallaway *et al.* 2013). During 2010-2018,  $N_t$

ranged 10,987-22,415 (Fig. 1), with its lowest in 2014 and highest in 2017, all of which were below predicted levels (Caillouet 2014; Caillouet *et al.* 2015b, 2016, 2018; Crowder & Heppell 2011; Dixon & Heppell 2015; Gallaway *et al.* 2013, 2016a, 2016b; Kocmoud *et al.* 2019; NMFS *et al.* 2011; NMFS & USFWS 2015). This represented a major setback in Kemp's ridley nesting (Caillouet *et al.* 2016). Deepwater Horizon Natural Resource Damage Assessment Trustees (2016) concluded that the oil spill was unlikely to have had a direct impact on Kemp's ridley nesting in 2010, but could have contributed to reduced numbers of nests in subsequent years through direct and indirect pathways. Gallaway *et al.* (2016b) estimated there were 61,330 Kemp's ridley deaths in 2010, of which they attributed 5% to incidental mortality in shrimp trawls, 19% to natural causes of mortality, and 76% to undetermined anthropogenic causes of mortality other than shrimp trawling. Various hypotheses have been put forward to explain the 2010-2018 nesting setback, but none have been confirmed with certainty (Gallaway *et al.* 2016a, 2016b; Caillouet *et al.* 2018; Kocmoud *et al.* 2019). Despite the nesting setback,  $h_t$  ranged 291,268-1,025,027 during 2000-2018, exceeding the 300,000-hatchling threshold established by NMFS *et al.* (2011) in all but one (2001) of the last 19 years (Fig. 1). Maintenance of such high levels of  $h_t$  for almost 2 decades demonstrates the dedication of Mexico to its on-beach conservation interventions in Tamaulipas and their efficacy.

Neritic immature and adult sea turtles are subject to compensatory density-dependent functions (National Research Council 2010). Lowered per capita availability of food for neritic Kemp's ridleys can reduce somatic growth rates, increase age at sexual maturity (ASM), and reduce body condition of adults and their ability to migrate to nesting beaches; it can reduce the ability of adult females to develop eggs and nest, as well as increase inter-nesting and remigration intervals (Bjorndal *et al.* 2014; Caillouet 2014; Caillouet *et al.* 2016, 2018; Gallaway *et al.* 2016b; Avens *et al.* 2017; Omeyer *et al.* 2017; Craven *et al.* 2019; Kocmoud *et al.* 2019). It is unlikely that oceanic stage Kemp's ridleys compete with conspecific neritic immatures or adults for food and other resources, but likely that neritic immatures and adults do compete for food and other resources. It is unlikely that availability of nesting beaches

has limited Kemp's ridley population growth; as the population increased, nesting spread within and beyond Tamaulipas, arribada size increased on beaches of Tamaulipas and elsewhere, and arribada nesting is the norm (Márquez *et al.* 1999; Jiménez-Quiroz *et al.*

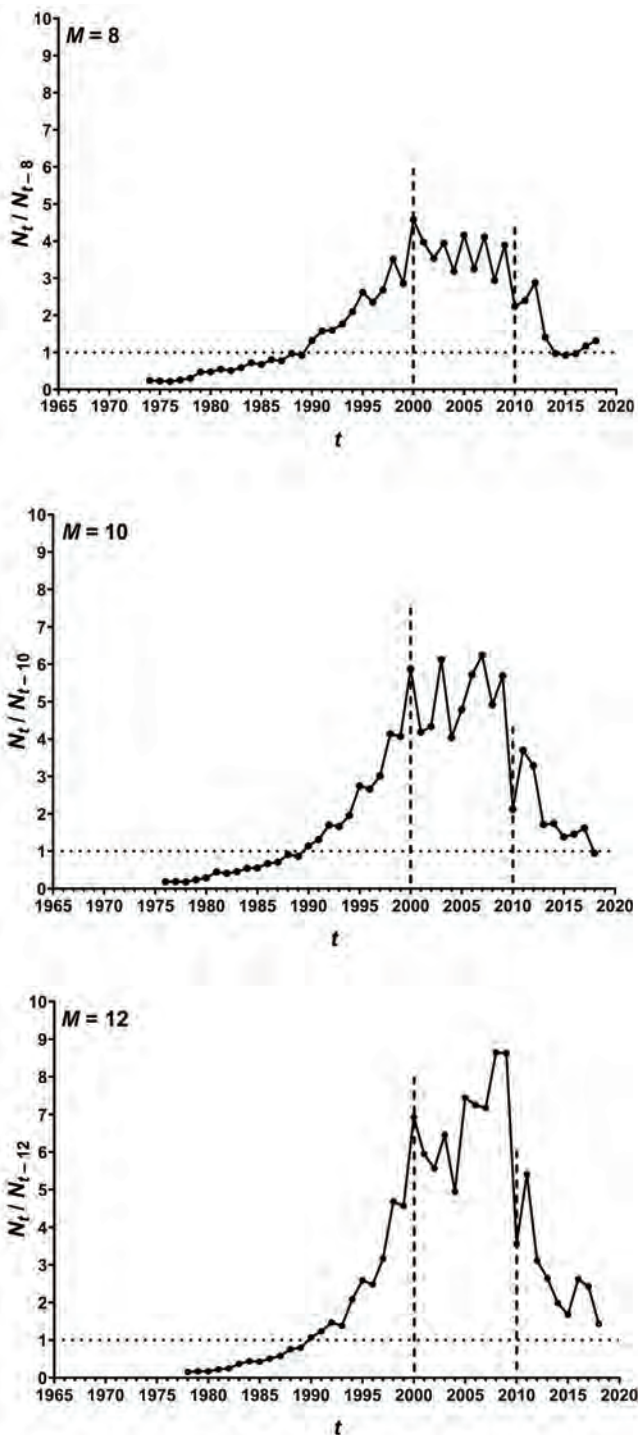
2003; Heppell *et al.* 2005, 2007; Márquez-M. *et al.* 2005; NMFS *et al.* 2011; Márquez-Millán *et al.* 2014). However, average  $h_t/N_t$  has been declining since it peaked in 1989, and its cause has not been determined (Caillouet 2014; Caillouet *et al.* 2016).

Density independence has been assumed in most modeling of the Kemp's ridley population (TEWG 1998, 2000; NMFS *et al.* 2011; Heppell *et al.* 2005, 2007; Coyne & Landry 2007; Crowder & Heppell 2011; Gallaway *et al.* 2013, 2016a; Dixon & Heppell 2015; NMFS & USFWS 2015). However, density-dependent effects on various vital (demographic) rates were evident before 2010 (Heppell *et al.* 2005, 2007; Caillouet 2014; Caillouet *et al.* 2016, 2018; Gallaway *et al.* 2016b; Shaver *et al.* 2016; Avens *et al.* 2017). The regression model applied by Caillouet *et al.* (2018) showed that density dependence affected  $N_t$  before 1962 and after 2004, with an interval of density-independent changes in  $N_t$  in between. Caillouet *et al.* (2018) hypothesized that slowing of the rate of increase in  $N_t$  after 2004 was caused by a combination of declining carrying capacity for Kemp's ridleys due to degradation of the GoM ecosystem, exponential growth of the population, and declining per capita availability of food for neritic immatures and adults, including natural prey and scavenged discarded bycatch from shrimp trawling. Factors that could have contributed to declining per capita availability of food included intraspecific competition among neritic immatures and adults, their interspecific competition with other marine predators and scavengers, effects of fisheries for crabs, and reductions in discarded bycatch from the shrimp fishery (Gallaway *et al.* 2016b; Avens *et al.* 2017; Caillouet *et al.* 2018; Craven *et al.* 2019; Kocmoud *et al.* 2019). The more abundant loggerhead sea turtle (*Caretta caretta*) may also compete for food with Kemp's ridley (Hart *et al.* 2018; Lamont & Iverson 2018).

Caillouet *et al.* (2018) prompted my examination herein of four novel and simple rates of change calculated from  $N_t$ ,  $h_t$ , and  $H_t$ , where  $H_t$  is the cumulative annual count of hatchlings released from the nester-abundance-index beach (see Figure 7 in Caillouet *et al.* 2016), to determine whether these rates exhibited pre-2010 evidence of density dependence. The range in  $t$  for these calculations was 1966-2018. Because assumed ASM affects results of Kemp's ridley population models, I incorporated three different values (8, 10, and 12 years) for  $M$  (minimum ASM) in calculating some of these rates. The range in published estimates of ASM for wild Kemp's ridleys is 6.8-21.8 years (Snover *et al.* 2007; NMFS *et al.* 2011; Avens *et al.* 2017). Models applied by TEWG (1998) and Heppell *et al.* (2005) incorporated assumed ASMs of 8, 10, and 12 years. Each value of  $M$  was assumed constant over  $t$ , as is usually the case with ASM in various models (but see the review by Bernardo 1993). In modeling, increasing the value of ASM increases the number of cohorts of neritic immatures in the estimated population, because the number of cohorts in the oceanic stage is typically held constant (TEWG 1998, 2000; Heppell *et al.* 2005, 2007; Coyne & Landry 2007; Crowder & Heppell 2011; NMFS *et al.* 2011; Gallaway *et al.* 2013, 2016a, 2016b; Kocmoud *et al.* 2019).

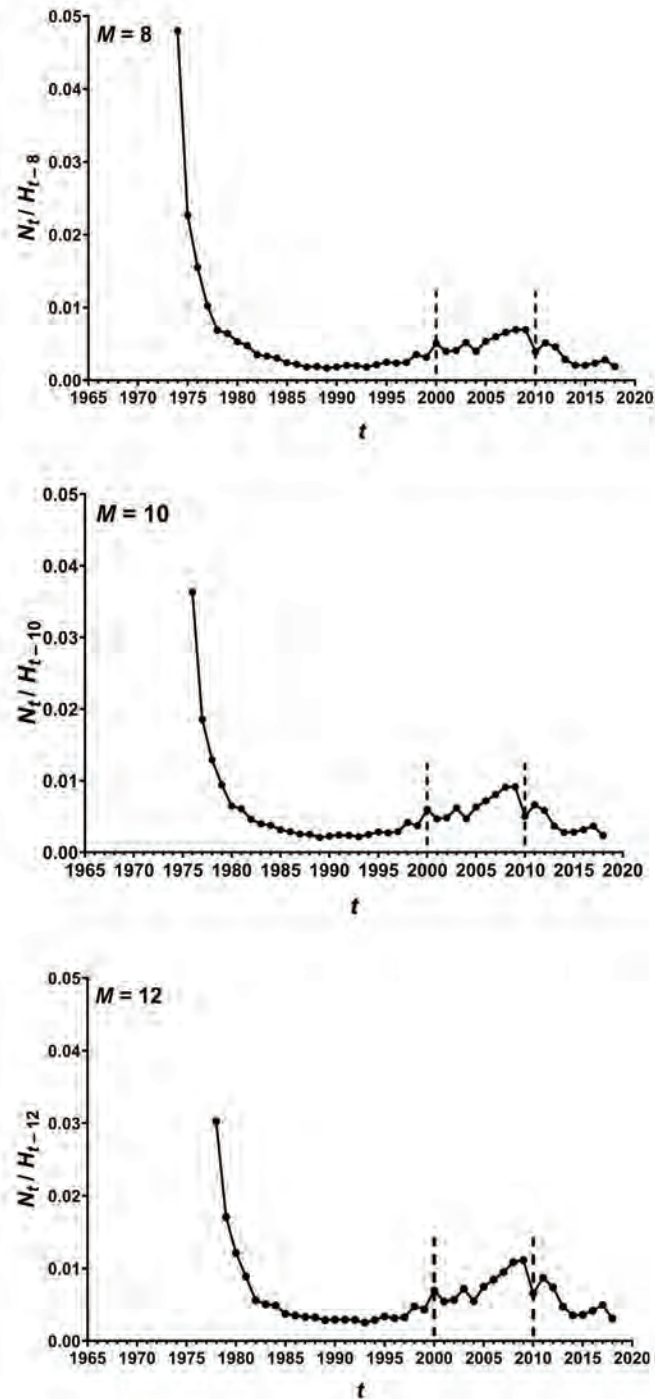
The four rates were:

- (1)  $N_t/N_{t-M}$  for  $M$  values of 8, 10, and 12 years;
- (2)  $N_t/H_{t-M}$  for  $M$  values of 8, 10, and 12 years;
- (3)  $h_t/h_{t-1}$ , the finite multiplication rate based on hatchlings released in each pair of consecutive years, and
- (4)  $h_t/H_{t-M}$  for  $M$  values 8, 10, and 12 years.



**Figure 2.** Trends in Kemp's ridley  $N_t/N_{t-M}$ , where  $t$  is calendar year,  $N_t$  is the annual number of nests (clutches of eggs laid) on the Tamaulipas, Mexico nester-abundance-index beach during 1974-2018, 1976-2018, and 1978-2018, for assumed minimum age at sexual maturity,  $M$ , of 8 years, 10 years, and 12 years, respectively. Dotted lines represent  $N_t/N_{t-M} = 1$ .

Regardless of assumed  $M$ , 2000 and 2009 were pivotal years for the trend in  $N_t/N_{t-M}$  as shown by substantial slowing of its trends after 2000 and again after 2009. Early values of  $N_t/N_{t-M}$  were below 1 for the three values of  $M$  (Fig. 2). Starting values of  $N_t/N_{t-M}$  were 0.244 for an  $M$  of 8 years, 0.185 for  $M=10$  years, and 0.154 for  $M$



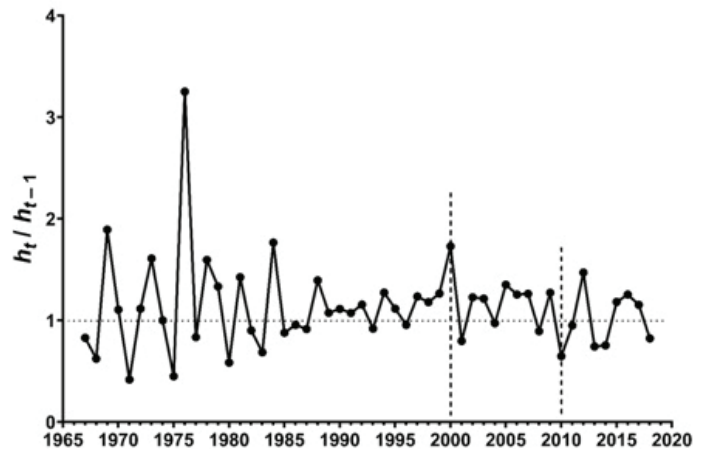
**Figure 3.** Trends in Kemp's ridley  $N_t/H_{t-M}$  where  $t$  is calendar year,  $N_t$  is annual number of nests (clutches of eggs laid) on the Tamaulipas, Mexico nester-abundance-index beach during 1974-2018, 1976-2018, and 1978-2018, for assumed minimum age at sexual maturity,  $M$ , of 8 years, 10 years, and 12 years, respectively, and  $H_{t-M}$  is cumulative annual number of hatchlings released on the nester-abundance-index beach during 1966-2010, 1966-2008, and 1966-2006, respectively.

$=12$  years. Through 2000,  $N_t/N_{t-M}$  increased least rapidly for  $M=8$  years, more rapidly for  $M=10$  years, and most rapidly for  $M=12$  years, because increases in  $M$  postponed the starting points for  $N_t/N_{t-M}$  thereby shortening the interval between starting years and 2000. Caillouet *et al.* (2018) plotted the time series of  $N_t/N_{t-1}$ , referring to it as the finite multiplication rate. Its highest level occurred in 2000 and its lowest in 2010 (Figure 1 B in Caillouet *et al.* 2018). The plot of residuals for the demographic model applied in the recovery plan (Figure 5 in NMFS *et al.* 2011) provided evidence of an inflection point in growth of  $N_t$  in 2000 when the largest positive residual occurred.

All values of  $N_t/H_{t-M}$  were below 1 (Fig. 3), because  $H_t$  was so much larger than  $N_t$  throughout the time series (Fig. 1), as expected. Starting values of the trends in  $N_t/H_{t-M}$  were highest (0.0480) for  $M=8$  years, intermediate (0.0363) for  $M=10$  years, and lowest (0.0302) for  $M=12$  years. All trends in  $N_t/H_{t-M}$  were steeply downward during the pre-1986 population decline. The pre-2010 minimum  $N_t/H_{t-M}$  was 0.00168 in 1989 for  $M=8$  years, 0.00206 in 1989 for  $M=10$  years, and 0.00252 in 1993 for  $M=12$  years. Regardless of assumed  $M$ , 2000 and 2009 were pivotal years for the trends in  $N_t/H_{t-M}$ . Caillouet *et al.* (2016) were the first to plot  $N_t/H_{t-M}$ , limiting it to  $M=10$  (see their Figure 7).

Variation in values of  $h_t/h_{t-1}$  was relatively wide during the first  $\approx 2$  decades, then narrowed through 2000 (Fig. 4). Interestingly, the highest value of  $h_t/h_{t-1}$  occurred in 1976, prior to the beginning of the US-Mexico Kemp's ridley restoration and enhancement program. Years 2000 and 2009 were pivotal for  $h_t/h_{t-1}$ , with  $h_t/h_{t-1}$  exhibiting a general decline with increased variability after 2000.

Starting values of the trends in  $h_t/H_{t-M}$  were lowest (0.81) for  $M=8$  years, intermediate (1.18) for an  $M=10$  years, and highest (1.57) for  $M=12$  years (Fig. 5). For all values of  $M$ , the trends in  $h_t/H_{t-M}$  were downward as the population declined; the downward trend of  $h_t/H_{t-M}$  was least steep for an  $M=8$  years, intermediate for  $M=10$  years, and most steep for  $M=12$  years. Interestingly, the pre-2010 minimum  $h_t/H_{t-M}$  was 0.00168 in 1989 for  $M=8$  years, 0.00206 in 1989 for  $M=10$  years, and 0.00252 in 1993 for  $M=12$  years, although minimum  $N_t$  occurred in 1985. Regardless of assumed  $M$ , 2000 and 2009 were pivotal years for the trend in  $h_t/H_{t-M}$ , with drops in 2010 marking the beginning of the nesting setback.



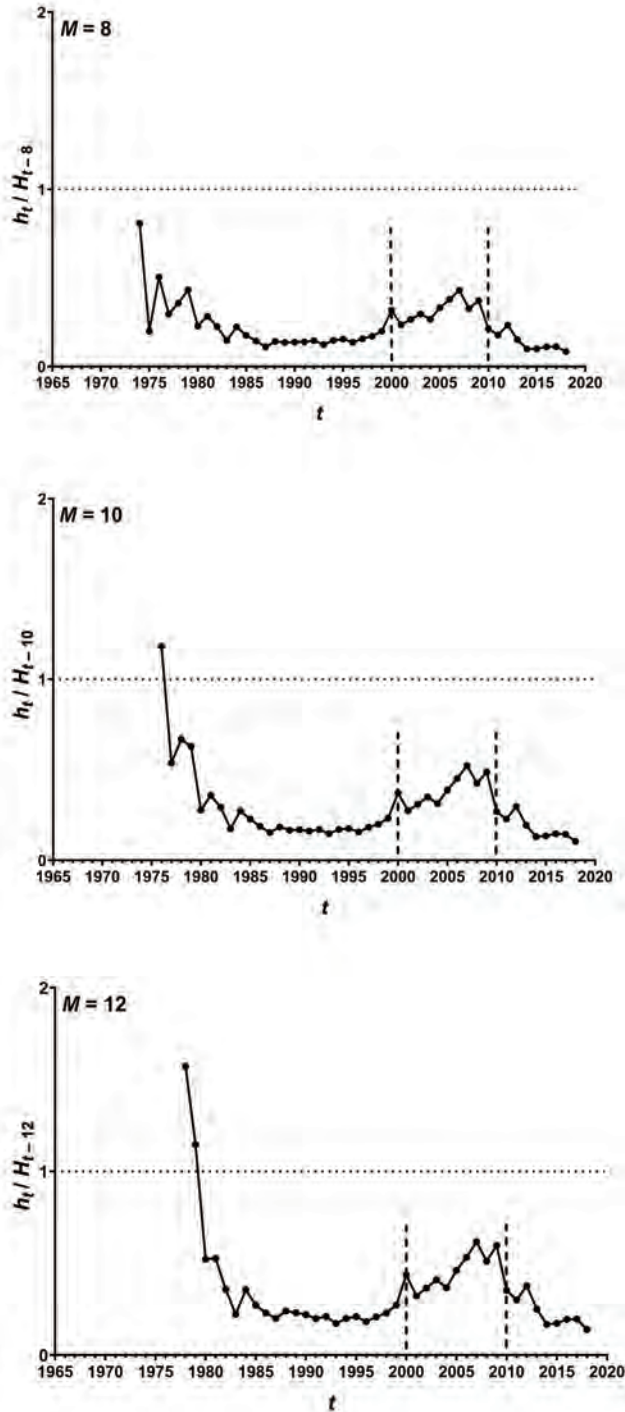
**Figure 4.** Trend in Kemp's ridley  $h_t/h_{t-1}$ , where  $t$  is calendar year,  $h_t$  is annual number of hatchlings released on the Tamaulipas, Mexico nester-abundance-index beach during 1967-2018. The horizontal dotted line represents  $h_t/h_{t-1} = 1$ .

Given that density dependence appears to have begun reducing the rate of growth of the Kemp's ridley population around year 2000, I hypothesize that annual numbers of neritic immatures became excessive around that year. I recommend that age-structured

modeling that incorporates estimates of annual mortality attributable to shrimp trawling in the GoM (e.g., Gallaway *et al.* 2013, 2016a, 2016b) be conducted to estimate annual numbers of adults and neritic immatures in the population during 1985-2018. For such modeling, data covering Mexico's shrimp trawling in the GoM should be acquired and combined with data covering US shrimp trawling in the GoM, for purposes of estimating total annual Kemp's ridley mortality attributable to shrimp trawling within the entire GoM. The 1985-2018 trend in the annual quotient calculated by dividing estimated annual number of adults by estimated annual number of neritic immatures can then be examined. If this annual quotient initially increased then later declined, the decline would suggest density-dependent limitation of population growth and show when it began developing. This approach is consistent with earlier modeling that estimated the potential of experimental reintroduction of Kemp's ridley nesting to Padre Island National Seashore and use of TEDs to contribute to Kemp's ridley population growth (Caillouet *et al.* 2015b; Heppell *et al.* 1996, 2005, 2007; Heppell & Crowder 1998; NMFS *et al.* 2011; NMFS & USFWS 2015; Shaver & Caillouet 2015; TEWG 1998, 2000). I welcome the application of other types of age-structured modeling to estimate annual numbers of adults and neritic immatures for use in calculating the suggested quotient and its trend beginning with 1985.

Pritchard (2007) pondered the possibility that the unstated goal of producing "as many turtles as possible" should be abandoned, "not only because natural population constraints will eventually be felt on the feeding grounds but also because there is almost certainly some level of density of an arribada at which the sheer number of turtles is counterproductive, leading to degradation of the beach and massive, although accidental, destruction of eggs laid by previous nesters". Natural population constraints on the feeding grounds seem more likely to have begun limiting the Kemp's ridley population's growth rate than arribada density, because most nests have been protected in on-beach hatcheries, beginning in 1978 (Caillouet 2006).

In all modeling of Kemp's ridley population dynamics to date, additions from immigration and losses from emigration have been ignored. However, the proportion of the population retained within the GoM is much greater than that in the Atlantic (NMFS *et al.* 2011; Putman *et al.* 2013; NMFS & USFWS 2015). I assume that Kemp's ridley immigration represents the return of neritic stage turtles from the North Atlantic Ocean (NAO) to the GoM, and emigration represents transport of oceanic stage turtles into the NAO combined with movement of neritic stage turtles from the GoM to the NAO. Migration distances from the NAO to western GoM nesting beaches are longer than those from within the GoM. Annually, the oceanic stage is much more abundant than the neritic stage, so losses to the NAO likely exceed gains by the GoM. The total number of GoM tag returns for Kemp's ridleys tagged along the US east coast is low, although most of them were documented for Tamaulipas nesters (Caillouet *et al.* 2015b). It is time for future Kemp's ridley population models to incorporate metrics of emigration and immigration, based on dispersal in the oceanic stage and examinations of available catch-mark-recapture and tracking data for neritic stage turtles, to determine whether there is a net loss to the NAO, and if so to estimate its magnitude. Notwithstanding possible net loss from the population through emigration into the NAO, nestings on the US east coast appear to be increasing, and may someday reach levels important to sustaining the population, just



**Figure 5.** Trends in Kemp's ridley  $h_t/H_{t-M}$  where  $t$  is calendar year,  $h_t$  is annual number of hatchlings released on the Tamaulipas, Mexico nester-abundance-index beach during 1974-2018, 1976-2018, and 1978-2018, for assumed minimum age at sexual maturity,  $M$ , of 8 years, 10 years, and 12 years, respectively, and  $H_{t-M}$  is cumulative annual number of hatchlings released on the nester-abundance-index beach during 1966-2010, 1966-2008, and 1966-2006, respectively. Horizontal dotted lines represent  $h_t/H_{t-M} = 1$ .

as nestings in the GoM in locations other than Tamaulipas provide “safety nets” for the species.

Heppell *et al.* (2007) and Wibbels & Bevan (2016) suggested that demographic criteria for delisting Kemp’s ridley may be unachievable. Consideration should be given to adding a recovery criterion related to achieving an annual population age-structure similar to that in 1947; *i.e.*, one with a much higher proportion of adult females than currently exists (Caillouet *et al.* 2018). A challenge is that the effects of any changes in conservation interventions will not be detectable via the  $N_t$  metric for approximately a decade. If reduced GoM carrying capacity for Kemp’s ridleys is currently the dominant factor limiting population growth, then ongoing efforts to restore the GoM ecosystem may mitigate its effects (see Caillouet *et al.* 2018; National Academies of Sciences, Engineering, and Medicine 2017; Peterson *et al.* 2011). In the interim, continued monitoring of  $N_t$  and  $h_t$  will make it possible to observe the effects of *status quo* maintaining or increasing  $h_t$  (NMFS *et al.* 2011). Consistent with this *status quo* approach was the suggestion by Caillouet *et al.* (2016) that the most expedient way to restore Kemp’s ridley population growth toward recovery would be to translocate more clutches to protective corrals, leaving fewer *in situ*, but I would recommend against it at this time.

If annual numbers of neritic immatures in the population are already excessive and preventing population recovery as defined by NMFS *et al.* (2011), it would seem prudent to begin reducing numbers of neritic immatures by reducing annual numbers of hatchlings released from Tamaulipas beaches. This could be achieved by leaving more clutches *in situ* without protection (NMFS *et al.* 2011; TEWG 1998, 2000). This could free some of the personnel and resources now devoted to collecting, translocating, and protecting clutches, to focus on researching (1) past, present, and future annual proportions of putative neophyte nesting females (Caillouet 2014), (2) cause(s) of the post-1989 decline in  $h_t/N_t$  (Caillouet 2014), (3) past, present, and future annual carapace length-frequency distributions of nesting females (Caillouet 2014), (4) past, present, and future health and body condition of nesting females (5) past, present, and future remigration intervals (via catch-mark-recapture) of nesting females (Gallaway *et al.* 2016b; Kocmoud *et al.* 2019), (6) sampling methods that ensure accurate counts of nesting females (NMFS *et al.* 2011; Rees *et al.* 2018) and hatchlings released, (7) life-long tags or marks for mass-tagging cohorts of hatchlings (Caillouet & Higgins 2015), (8) past, present, and future annual sex ratios of hatchlings, and (9) detection of tags or marks on nesting females (external and internal). Consideration should also be given to updating and modifying the bi-national recovery plan (Caillouet 2006; Caillouet *et al.* 2015a), including the demographic criteria for downlisting and delisting. Whether or not the analyses and modeling recommended herein are conducted, continued on-beach conservation interventions (at a level to be determined) and monitoring on the coast of Tamaulipas are essential to Kemp’s ridley population recovery within the GoM, and they are required to maintain and enhance the secondary and tertiary nesting colonies in Veracruz and Texas that contribute to the population’s diversity and resilience (NMFS *et al.* 2011; NMFS & USFWS 2015; Tecolutla Turtle Project 2018).

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# The Longest Migratory Distance Recorded for a Loggerhead Nesting in Greece

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Mediterranean loggerhead turtles nest mainly in eastern Mediterranean and particularly in Greece, Turkey, Cyprus and Libya with Greece hosting the largest number of nests (Casale *et al.* 2018). As juveniles, Mediterranean loggerheads may migrate into the western Mediterranean as evidenced by genetic markers (Clusa *et al.* 2014). However, the majority of adult females, as revealed by flipper tag returns and satellite tracking, remain within the eastern Mediterranean (Margaritoulis 1988, Margaritoulis *et al.* 2003, Margaritoulis & Rees 2011, Zbinden *et al.* 2011, Schofield *et al.* 2013, Patel *et al.* 2015, Snape *et al.* 2016).

In 2013 ARCHELON carried out a satellite tracking project on seven loggerhead turtles in the vicinity of Mesolonghi Lagoon, Greece (38.323 °N, 21.357 °E), a foraging area for loggerhead turtles, but with the occasional presence of green turtles reported. This paper presents the results of one satellite tagged loggerhead turtle that was re-observed on a nesting beach two months after the transmitter had ceased operation.

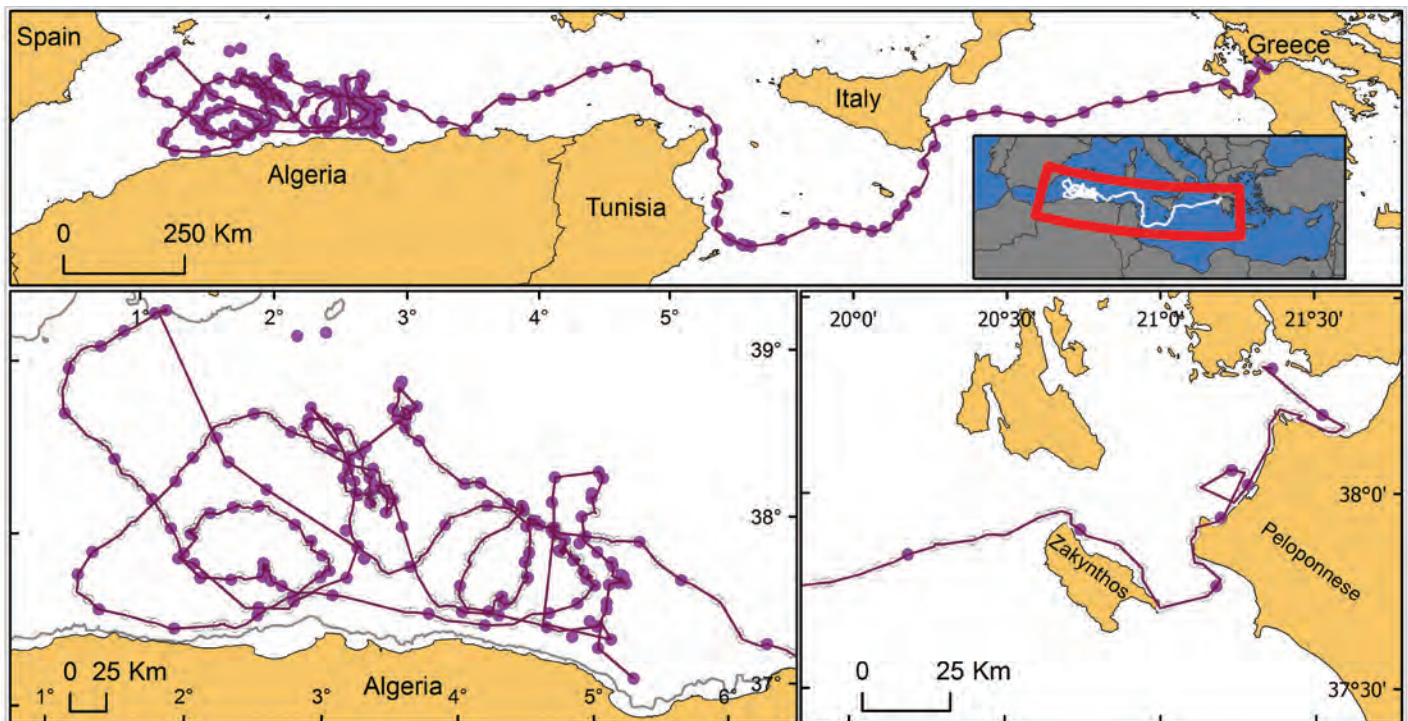
Turtles in July 2013 were captured by turtle rodeo technique from the shallow waters in and adjacent to the Mesolonghi Lagoon. Curved carapace length, notch to tip (CCL), and straight carapace length, notch to tip (SCL), were measured for each turtle, any existing injuries were recorded, and turtles were flipper tagged with

a single Monel metal tag on the trailing edge of each front flipper. Turtles were equipped with Kiwisat 202 Platform Transmitter Terminals (satellite tags) attached to the carapace with 2-part epoxy (see cover photo). Tracking route data were filtered to retain the best location per day with Argos LC quality ordered from best to worst; 3, 2, 1, 0, A, B. If there were two locations of equal best quality in one day, then the one closest to 12:00 UTC was retained.

Complementing this work, ARCHELON annually monitors several loggerhead nesting beaches in Greece during nesting season, including night time patrols that include flipper tagging post-nesting individual turtles (Margaritoulis 1988).

One female loggerhead turtle named Reggina was captured, close to Mesolonghi Lagoon, on 15 July 2013. She exhibited a healed injury to the anterior left margin of her carapace but this did not affect the collection of accurate length measurements (Table 1). She bore no evidence of having been flipper tagged previously.

Upon release on the same date of its capture, Reggina departed the Mesolonghi area immediately, heading south then west, passing along the east and north side of Zakynthos Island, *en route* to Italy. From Italy she continued through Maltese waters before reaching Tunisia and proceeded further into the western Mediterranean, reaching the deep seas off Algeria and at one point the turtle



**Figure 1.** Reggina's migration from Mesolonghi Lagoon to the western Mediterranean. Upper panel: Regina's complete migration from east to west with inset for regional context. Lower left panel: long-term oceanic looping undertaken by Reggina until the transmitter ceased functioning. 500 m isobath shown as grey line. Lower right panel: Origin of the migration from Mesolonghi Lagoon (upper right) including Reggina's passing of Zakynthos Island where she would return to nest three years later.

Date	CCL	SCL
15/07/2013	77.0	72.1
21/07/2016	78.0	74.0

**Table 1.** Carapace lengths (cm) for Reggina from first and last observation.

approached the Spanish island of Ibiza, around 1800 km from where she was tagged (Fig. 1).

She undertook extensive oceanic circling between 5 September 2013 and 24 February 2014 (172 days) when transmissions stopped. Average sea depth experienced by the turtle during this time was 2,614 m (SD=467, range=333-2,876 m, n=150 days). Her last location was received on 24 February 2014, only 15 km from coast of Algeria but still in water 1,440 m deep.

Reggina was next observed while nesting on Sekania Beach, Zakynthos Island, Greece, on 21 July 2016, by researchers working on ARCHELON's long-term nesting beach tagging project. While nesting, her carapace was again measured (Table 1) and indicated she had grown at least 1 cm in the intervening three years. The turtle was identified by her two flipper tags and characteristic left-side injury. There was no observable evidence of the satellite tag attachment on the carapace, with the transmitter and epoxy having been shed at some point after transmissions ceased.

We do not know whether the turtle was an adult when it was first encountered in Mesolonghi Lagoon in 2013 and equipped with its satellite tag, but this track represents the furthest distance from its nesting beach that a turtle known to nest in Greece has been tracked (>1700 km). Other published records of turtles nesting in Greece do not include any migratory distances greater than 1300 km from the nesting area; these include all known flipper-tag returns as well as all published satellite tracks (Margaritoulis 1988, Margaritoulis *et al.* 2003, Margaritoulis & Rees 2011, Zbinden *et al.* 2011, Schofield *et al.* 2013, Patel *et al.* 2015). Additionally, one nesting female from Kyparissia Bay, west Peloponnese (90km SE from Zakynthos), has been tracked to her nesting beach from the northern Adriatic (Luschi *et al.* 2013) ca. 1,100 km away.

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# First Confirmed Hawksbill Nesting on the Pacific Coast of Guatemala

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The first confirmed nest made by a hawksbill sea turtle (*Eretmochelys imbricata*) on the Pacific coast of Guatemala occurred during the night of 22 July 2018 near the village of Madre Vieja (13.91103° N, 90.56161° W), 8 km west of the touristic resort of Monterrico. Local egg collectors Claudio and Estuard Montepeque were the first to encounter the turtle and were struck by its size and the fact that it took over 3 hours to lay its eggs. They were joined by one of us (SI), who photo-documented the nesting (Fig. 1). The nesting turtle had a metal flipper tag in its left front flipper (GK254, Fig. 2). A total of 156 eggs were collected from the nest and transported to the El Banco Hatchery, 4 km to the east, for protected incubation. All but five of the eggs produced hatchlings, which were then released to the ocean.

SI contacted CM, who then contacted members of the Eastern Pacific Hawksbill Initiative (ICAPO) and the National Oceanographic and Atmospheric Administration (NOAA). According to these groups, the flipper tag #GK 254 was first applied to an adult female hawksbill turtle nesting in Bahía de Jiquilisco, El Salvador on 09 June 2014. This site is ~300 km from the nest deposited at Madre Vieja, which is the longest distance between nests ever recorded for a single hawksbill turtle in the eastern Pacific. This is also one of the few examples of multinational nesting in the region by an individual hawksbill turtle.

In Guatemala, juvenile hawksbills are occasionally reported as incidentally captured by fisherman in mangrove estuaries and in the ocean (Gaos *et al.* 2010; Brittain *et al.* 2012). Between 1982 and 2009, two hawksbill nests were reported in Pacific Guatemala (Gaos *et al.* 2010), although these records lacked photographic evidence. The nest described here is the first confirmed hawksbill nest on the Pacific coast of Guatemala. The discovery is extremely significant considering that all nests on the Guatemalan Pacific coast are laid by olive ridley sea turtles (*Lepidochelys olivacea*), with green turtles (*Chelonia mydas*) and leatherback turtles (*Dermochelys coriacea*) also nesting infrequently in the country. This documentation should help raise awareness about the presence of hawksbills in Guatemala, and hopefully will result in additional records of local nesting by this critically endangered species.

**Acknowledgements.** We thank Ingrid Yañez of ICAPO and Jeff Seminoff of NOAA for information on the tagging history of this hawksbill turtle.

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**Figure 1.** Nesting hawksbill sea turtle on the Pacific coast of Guatemala, in July 2018. The eggs were collected and transported to a protected hatchery for incubation.



**Figure 2.** Metal flipper tag with ID GK254 on the hawkbill turtle's left front flipper.

D. CHACÓN, C. DUEÑAS, M. LILES, G. MARIONA, C. MUCCIO, J.P. MUÑOZ, W.J. NICHOLS, M. PEÑA, J.A. SEMINOFF, M. VÁSQUEZ, J. URTEAGA, B. WALLACE, I.L. YAÑEZ & P. ZÁRATE. 2010. Signs of hope in the eastern

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# Necropsy of a Green Turtle (*Chelonia mydas*) and the Impacts of Plastic Pollution in Tioman Island, Malaysia

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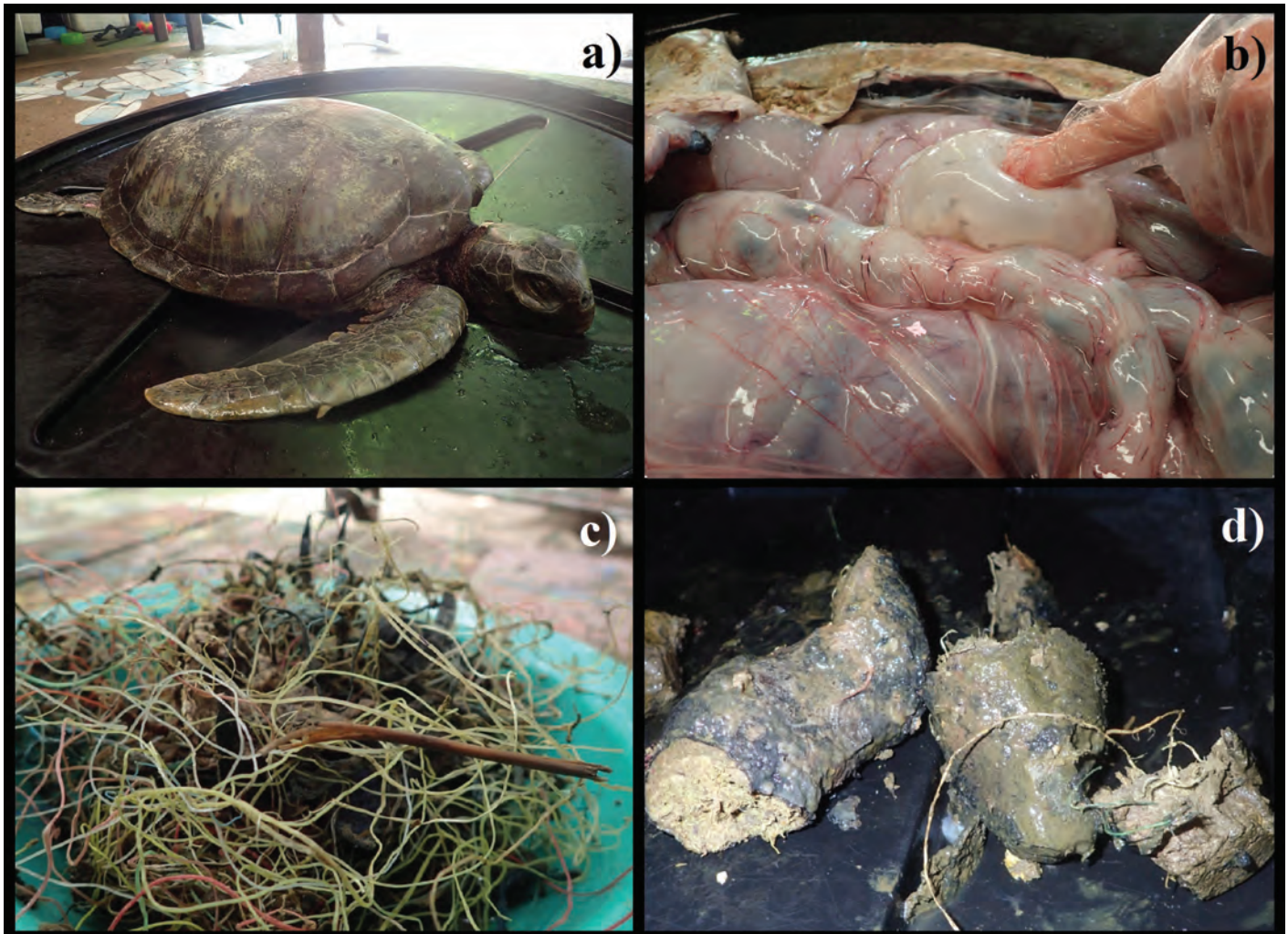
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On 08 June 2017, the Juara Turtle Project (JTP) received reports from Air Batang locals and officers of the Department of Fisheries (DoF) in Tioman Island, Malaysia, that a green turtle (*Chelonia mydas*) was displaying an abnormal behavior at the sea surface of coastal waters and was unable to dive. Despite efforts to save this individual, it was found in an extremely weak state, and it died few hours after being rescued. No external injuries were present, but there was little musculature around the flippers and neck, suggesting severe signs of starvation (Fig. 1a). The turtle measured 73.0 cm in curved carapace length and 59.5 cm in curved carapace width, and thus was identified as a juvenile.

At the JTP center, a necropsy was performed the same day of the turtle's death, revealing a massive obstruction in the digestive tract of the turtle. Gas pockets of considerable size formed in some parts of the intestines (Fig. 1b), which might have seriously restricted its

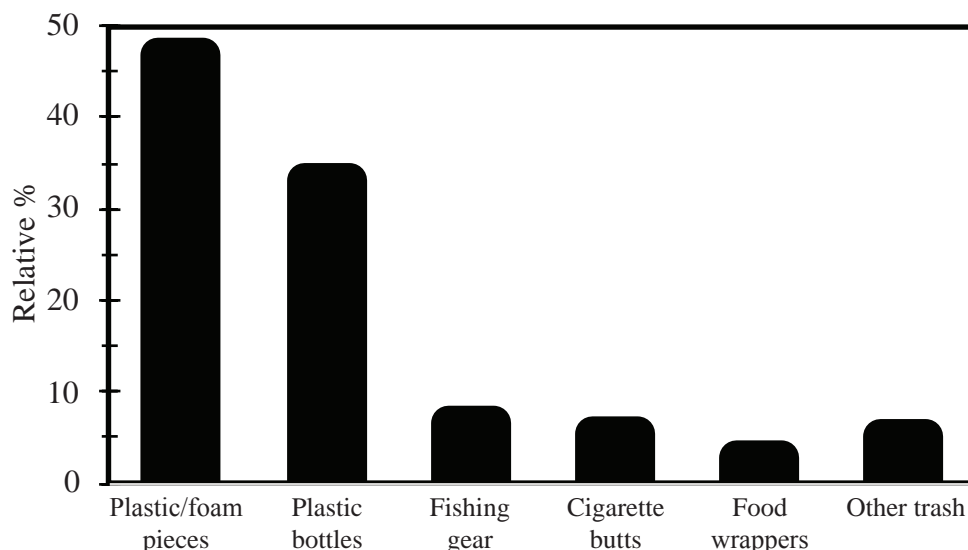
ability to dive. When its digestive content was examined, a mass of plastic, string, foam and monofilament lines was found embedded in a black layer of oil and bile (Figs. 1c, 1d). In total, 75 fragments of monofilament lines (48%), 55 string pieces (35%), 20 pieces of rope (13%), 3 pieces of foam (2%) and 3 plastic fragments (2%) were recovered. A video describing the whole process can be found online ([www.youtube.com/watch?v=Q4RQ\\_xH0Y4k](http://www.youtube.com/watch?v=Q4RQ_xH0Y4k)).

This is the first documented case of a sea turtle death in Tioman Island associated with plastic ingestion. However, the mortality of marine life due to debris ingestion is well documented worldwide (Mrosovsky *et al.* 2008; Bernardini *et al.* 2018; Germanov *et al.* 2018; Wilcox *et al.* 2018). Tioman Island has four nesting beaches for green turtles and hawksbill turtles (*Eretmochelys imbricata*), with an average of 60 nests per year. Moreover, the island also supports a population of resident juvenile green and hawksbill turtles that



**Figure 1.** A. Turtle before the necropsy. The area surrounding the eyes suggests signs of starvation. B. Gas pocket in the blocked digestive tract. C. Detail of contents recovered from the digestive tract after rinsing. D. Contents before rinsing.





**Figure 2.** Trash items collected from beach cleanups in Mentawak and Nayak (2018).

use the coastal area around the island as a feeding ground (~50 individuals identified). For juvenile sea turtles, just one gram of debris can lead them to death (Guimaraes-Santos *et al.* 2015).

From mid-September to early November 2018, JTP and volunteers organized beach clean-ups in Mentawak Beach and Nayak Bay, two locations where sea turtles are often spotted feeding in the nearshore waters. In total, 224.5 kg of trash was removed. Of the 7962 items identified, most were plastic/foam pieces (47%) and plastic bottles (33%), followed by other trash (7%), fishing gear (5%) and cigarette butts (3%) (Fig. 2). All the items collected were classified using the app *CleanSwell* developed by the Ocean Conservancy ([www.oceanconservancy.org](http://www.oceanconservancy.org)). Because most of the trash found during beach cleanups had washed in from the ocean, this reflects the potential threats of plastic on turtles around their feeding grounds. Given this, JTP has started a “plastic-free” campaign in Juara Village in Tioman Island in 2019 to reduce single-use plastic. We plan to provide stainless steel straws as well as re-usable tote bags to local businesses to reduce single-use plastic consumption in the village. This could be the cornerstone of promoting eco-friendly initiatives within the community.

Although the single case presented here constitutes only one example of the impacts of plastic debris ingestion on sea turtles, we suspect that this is not an isolated case in the area. Ongoing assessment on the impact of marine debris on the turtle populations nesting and feeding at Tioman Island is essential, and will reveal what percentage of mortality is due to plastic entanglement or ingestion.

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# Blood Cholesterol as a Biomarker of Fibropapillomatosis in Green Turtles

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Fibropapillomatosis (FP) is a neoplastic disease affecting turtles, especially the green turtle *Chelonia mydas* (Linnaeus 1758). The prevalence of FP has increased dramatically in the last two to three decades, and is now considered an emerging panzootic disease, threatening the survival of sea turtles due to mortality associated with the extensive damage that can be caused by tumors (Duarte *et al.* 2012). FP is a disease characterized by the presence of one or more potentially debilitating tumors (external and/or internal), affecting mainly juvenile turtles (Work *et al.* 2003). Although most tumors appear to be benign, their size, location and number can impair basic functions such as swimming, vision, feeding and respiration, and may cause malfunction of internal organs (Foley *et al.* 2005). According to Herbst (1994), turtles with tumors also are more susceptible to entanglement in fishing nets than those without tumors.

Considering the alarming increase in the incidence of cutaneous FP in green turtles worldwide, blood profiles of healthy and sick individuals from different regions have been analyzed as an attempt to assess the possible causes and consequences of FP (Aguirre *et al.* 1995; Work & Balazs 1999; Aguirre & Balazs 2000; Swimmer 2000; Rossi *et al.* 2009). In this context, hematology and blood biochemistry are considered useful diagnostic tools to assess and monitor the health and physical condition of turtles (Aguirre & Balazs 2000; Labrada-Martagón *et al.* 2010). These tools have been employed as indicators of physiological disturbances associated with a variety of diseases (Swimmer 2000; Whiting *et al.* 2007), stress (Knotková *et al.* 2005), and exposure to contaminants (Keller *et al.* 2004).

Changes in several physiological and biochemical parameters have been reported to be associated with FP in green turtles. They include anemia, immunosuppression, hypoproteinemia, hypoalbuminemia, hypoglycemia, uremia, electrolyte imbalance, increased activity of liver enzymes, low levels of cholesterol and triglycerides, propensity to acquire systemic bacterial infections, and alterations in the number of white blood cells (Foley *et al.* 2005; Work & Balazs 1999; Aguirre & Balazs 2000; Santos *et al.* 2015). Furthermore, previous studies have reported a relationship between the hematological status of turtles and the severity of tumors (Work & Balazs 1999; Santos *et al.* 2015). According to Balazs (1991), tumor scores reflect the spectrum of severity of FP in green turtles. In advanced stages of the disease, clinical tests usually indicate acidosis, imbalance in the ratio between calcium and phosphorus concentrations, anaemia, hypoproteinemia paralleled by hypoglobulinemia and hypoalbuminemia, hypoglycemia, uremia, and increased activity of liver enzymes (Aguirre *et al.* 1995; Work & Balazs 1999; Aguirre & Balazs 2000; Santos *et al.* 2015). Immunosuppression may also occur paralleled by bacteremia (Work *et al.* 2001, 2003).

However, these reported alterations are generalized responses to stressors and clear evidence of a primary response that could be

used as a reliable biomarker of FP in green turtles is still lacking. This is likely because blood parameters in sea turtles can be affected by several intrinsic and extrinsic factors (Aguirre *et al.* 1995). For example, the wide reference ranges for many biochemistry markers reported in green turtles with FP could be associated with factors such as gender or body size of the specimens, both of which are known to affect biochemistry. Previous studies have been performed on individuals of both sexes and of a wide range of body sizes (Work & Balazs 1999; Aguirre & Balazs 2000;), which could hamper the identification of a potential and reliable biomarker of the disease.

The coastal zone of the southern Atlantic Ocean is an important feeding area and habitat for the development of juvenile green turtles. In Brazil, the first case of FP in green turtles was reported in 1986 by the Marine Turtle (TAMAR) Project (Baptistotte *et al.* 2005). Since then, an increase in the prevalence of the disease has been reported in several studies across the TAMAR project region. For example, Mehnert *et al.* (2001) reported an increase along the Brazilian coast between 1990 and 1999 and the prevalence of FP in juvenile green turtles from the Ubatuba coastal region of Brazil rose from 0 to 24% in the 12 years from 1986 to 1998 (Rossi *et al.* 2009).

The aim of the present study is to identify a primary biochemical response that could be used as a potential biomarker of FP for use in future evaluation and monitoring of health status of immature and juvenile green turtles found in coastal waters of the southern Atlantic Ocean.

Green turtles were captured using purse seine and scuba diving activities from January 2011 to March 2012 in the TAMAR Project area (23°26'S, 45°05'W, Ubatuba, São Paulo State, southeastern Brazil). The turtles that ended up trapped in the purse seines were used in the study; in addition, turtles were captured swimming freely to complete the sampling. Turtles were transferred to the TAMAR Project facilities for blood sample collection and physical examination, as described below. Blood samples (5 – 10 ml) of 36 green turtles (*C. mydas*) were collected by puncture of the dorsal cervical sinus using disposable 10-ml syringes with 25 x 7-gauge needles. This procedure is considered a minimally invasive technique (Owens & Ruiz 1980). Blood samples were immediately transferred to harvesting tubes without anticoagulant. All procedures were performed under a permit of the Brazilian Ministry of Environment (permit # 25829-2 SisBio/ICMBio/MMA).

After blood sampling, each turtle was subjected to a visual examination, including evaluation of general physical condition and the presence of external FP tumors. Therefore, the presence of internal tumors in green turtles assessed as clinically normal cannot be ruled out. Considering that levels of hematological and serum biochemical parameters may differ according to the severity of the tumors (Santos *et al.* 2015; Hiramata *et al.* 2014), green turtles

were grouped and analyzed according to this condition (score 0: non-afflicted with FP; score 1: lightly afflicted with FP; score 2: moderately afflicted with FP; and score 3: heavily afflicted with FP), as described by Work & Balazs (1999).

After visual examination, curved carapace length (CCL; to the nearest 0.1 cm) was measured. Green turtles were then tagged on their front flippers, using metal Inconel style flipper tags provided by the TAMAR Project, placed in the center of the first or second scale proximal to the body of the turtle; turtle's with FP score 0 were immediately released close to the site of capture. Individuals afflicted with FP had their tumors removed surgically and were maintained for some time for observation and recovery. Once deemed sufficiently recovered, they were released near their site of capture.

Immediately after collection, sampled blood was divided into 2 tubes, one with heparin and one without anticoagulant. Whole blood was transferred to duplicate microcapillary tubes and centrifuged for 5 min using a microhematocrit centrifuge (Spin 1000, Microspin, Brazil). Hematocrit value was expressed as the average percentage value observed between the two hematocrit capillary tubes. Heparinized blood was used for the total leukocyte (WBC) and red blood cells (RBC) counts, which were performed using a Neubauer chamber. For each sample, two blood smears were also prepared, one fixed in methanol, and one stained with Wright-Giemsa stain (Campbell 2014). Differential leukocyte counting (heterophil, lymphocyte, eosinophil and monocyte) was performed manually; cells were identified using data reported in the literature regarding the morphology of sea turtle cells (Casal & Orós 2007; Zhang *et al.* 2011; Acevedo *et al.* 2012).

Immediately after the hematocrit analysis, the remainder of the whole blood was centrifuged at 1,800 x g for 5 min (CentriBio 80-2B, CentriBio, China). There was no visual evidence of hemolysis. Serum obtained was transferred into cryogenic vials kept on dry ice, transferred to the laboratory, and stored in an ultrafreezer (-80 °C) until analysis.

Serum biochemical parameters analyzed included cortisol, glucose, cholesterol, triglycerides, uric acid, urea, creatinine, total protein, albumin, globulin, bilirubin (total, direct and indirect), sodium, potassium, magnesium, chloride, calcium, and phosphorus concentration, as well as alanine aminotransferase (ALT),

aspartate aminotransferase (AST), alkaline phosphatase (ALP), gamma-glutamyl transpeptidase (GGT), creatine kinase (CK), lactate dehydrogenase (LDH), and 3-hydroxy-3-methylglutaryl-CoA reductase (HMGR) activity. Additionally, serum testosterone concentration was measured for sex identification (Bolten & Bjorndal 1992; Owens 1997).

Commercial reagent kits used to perform the serum biochemistry analyses were purchased from Labtest Diagnóstica (Lagoa Santa, MG, Brazil) and Sigma-Aldrich (St. Louis, MO, USA). Analyses were performed using a flame photometer (Micronal, Campo Grande, MS, Brazil), an automated Roche Cobas Mira Classic Chemistry Analyzer (Roche Molecular Systems, Branchburg, NJ, USA) and a microplate reader (Victor, PerkinElmer, Waltham, MA, USA).

For all parameters, data were expressed as  $X \pm SE$ . For each parameter, data normality was checked by the normal probability plot of raw residuals while homogeneity of variances was verified using the Cochran *C* test. Data on CCL, hematocrit, eosinophils, basophils, glucose, uric acid, potassium, magnesium, phosphorus, ALT, ALP, GGT, CK, LDH, and HMGR were mathematically transformed (decimal logarithmic transformation) to meet the analysis of variance (ANOVA) assumptions (data normality and homogeneity of variances). Mean values for all parameters were compared using one-way ANOVA followed by the Fisher LSD test. In all cases, the significance level adopted was 95% ( $\alpha = 0.05$ ). Mean values of HMGR activity and serum cholesterol concentration were significantly different among the groups of green turtles. Therefore, they were subjected to the Product-Moment correlation analysis. In all cases, the significance level adopted was 5% ( $\alpha = 0.05$ ) (Sokal & Rohlf 1995).

Among the 36 green turtles sampled, 31 individuals (CCL =  $40.3 \pm 1.6$  cm; BM =  $8.8 \pm 1.3$  kg) showed serum testosterone concentration  $<10$  pg/ml and were considered as being females (Bolten & Bjorndal 1992; Owens 1997). The CCL of these individuals ranged from 29.3 to 62.0 cm, which is in the range of sizes for immature and juvenile green turtles. Among the 31 green turtles analyzed in the present study, 14 individuals were non-afflicted with FP (score 0; CCL =  $36.8 \pm 1.2$  cm) while 17 green turtles had FP (external tumors). Among those with FP, 5 green turtles were lightly afflicted with FP (score 1; CCL =  $47.1$

Parameter	Tumor score			
	0 (n = 14)	1 (n = 5)	2 (n = 5)	3 (n = 7)
Hematocrit (%)	27.3 ± 2.1 <sup>a</sup>	28.2 ± 2.6 <sup>a</sup>	18.0 ± 4.0 <sup>a</sup>	20.8 ± 3.6 <sup>a</sup>
RBC (x 103/mm <sup>3</sup> )	363.0 ± 29.1 <sup>a</sup>	380.4 ± 53.1 <sup>a</sup>	488.0 ± 107.0 <sup>a</sup>	409.4 ± 128.8 <sup>a</sup>
WBC (x 103/mm <sup>3</sup> )	12.9 ± 2.1 <sup>a</sup>	11.2 ± 3.5 <sup>a</sup>	14.7 ± 8.4 <sup>a</sup>	10.6 ± 1.8 <sup>a</sup>
Heterophils (%)	53.7 ± 7.6 <sup>a</sup>	50.4 ± 6.5 <sup>a</sup>	33.3 ± 5.1 <sup>a</sup>	46.0 ± 11.8 <sup>a</sup>
Lymphocytes (%)	34.8 ± 6.3 <sup>a</sup>	38.8 ± 6.2 <sup>a</sup>	53.2 ± 6.3 <sup>a</sup>	34.2 ± 9.6 <sup>a</sup>
Eosinophils (%)	6.8 ± 3.8 <sup>a</sup>	4.4 ± 2.0 <sup>a</sup>	5.0 ± 2.0 <sup>a</sup>	14.2 ± 9.6 <sup>a</sup>
Monocytes (%)	2.7 ± 0.8 <sup>a</sup>	4.8 ± 0.9 <sup>a</sup>	5.2 ± 0.9 <sup>a</sup>	4.0 ± 1.8 <sup>a</sup>
Basophils (%)	2.2 ± 1.2 <sup>a</sup>	2.2 ± 1.7 <sup>a</sup>	3.3 ± 1.4 <sup>a</sup>	1.6 ± 1.6 <sup>a</sup>

**Table 1.** Hematological parameters in juvenile female green sea turtles with and without fibropapillomatosis (FP). Individuals were collected in coastal waters of the southern Atlantic Ocean (Ubatuba, southeastern Brazil) from January 2011 to March 2012. They were grouped according to the severity of tumors. Data are expressed as  $X \pm SE$ , with sample size in parentheses. Same letters indicate mean values are not significantly different ( $p < 0.05$ ).

Parameter	Tumor score			
	0 (n = 14)	1 (n = 5)	2 (n = 5)	3 (n = 7)
Cortisol (µg/dL)	0.46 ± 0.06 <sup>a</sup>	0.40 ± 0.00 <sup>a</sup>	0.40 ± 0.00 <sup>a</sup>	0.40 ± 0.00 <sup>a</sup>
Glucose (mg/dL)	88.4 ± 8.3 <sup>a</sup>	74.6 ± 7.8 <sup>a</sup>	95.2 ± 14.3 <sup>a</sup>	86.3 ± 10.5 <sup>a</sup>
Cholesterol (mg/dL)	153.1 ± 17.4 <sup>a</sup>	141.3 ± 22.9 <sup>a</sup>	68.8 ± 16.0 <sup>b</sup>	69.3 ± 15.8 <sup>b</sup>
Triglycerides (mg/dL)	53.1 ± 7.3 <sup>a</sup>	57.0 ± 8.1 <sup>a</sup>	45.0 ± 6.6 <sup>b</sup>	44.7 ± 5.4 <sup>a</sup>
Uric acid (mg/dL)	0.94 ± 0.17 <sup>a</sup>	0.94 ± 0.29 <sup>a</sup>	0.94 ± 0.38 <sup>a</sup>	0.81 ± 0.23 <sup>a</sup>
Urea (mg/dL)	65.1 ± 14.1 <sup>a</sup>	51.8 ± 20.1 <sup>a</sup>	90.6 ± 19.2 <sup>a</sup>	86.7 ± 20.9 <sup>a</sup>
Creatinine (mg/dL)	0.39 ± 0.06 <sup>a</sup>	0.35 ± 0.06 <sup>a</sup>	0.48 ± 0.07 <sup>a</sup>	0.34 ± 0.03 <sup>a</sup>
Protein (g/dL)	2.62 ± 0.28 <sup>a</sup>	3.25 ± 0.41 <sup>a</sup>	2.71 ± 0.56 <sup>a</sup>	2.61 ± 0.44 <sup>a</sup>
Albumin (g/dL)	1.22 ± 0.13 <sup>a</sup>	1.25 ± 0.11 <sup>a</sup>	0.97 ± 0.14 <sup>a</sup>	1.04 ± 0.14 <sup>a</sup>
Globulin (g/dL)	1.41 ± 0.20 <sup>a</sup>	2.02 ± 0.55 <sup>a</sup>	1.74 ± 0.44 <sup>a</sup>	1.57 ± 0.31 <sup>a</sup>
Total bilirubin (mg/dL)	0.067 ± 0.006 <sup>a</sup>	0.073 ± 0.015 <sup>a</sup>	0.052 ± 0.005 <sup>a</sup>	0.051 ± 0.010 <sup>a</sup>
Direct bilirubin (mg/dL)	0.028 ± 0.003 <sup>a</sup>	0.022 ± 0.004 <sup>a</sup>	0.022 ± 0.004 <sup>a</sup>	0.019 ± 0.003 <sup>a</sup>
Indirect bilirubin (mg/dL)	0.039 ± 0.004 <sup>a</sup>	0.051 ± 0.012 <sup>a</sup>	0.030 ± 0.003 <sup>a</sup>	0.033 ± 0.007 <sup>a</sup>
Sodium (mEq/L)	156.1 ± 3.1 <sup>a</sup>	144.4 ± 8.7 <sup>a</sup>	145.4 ± 2.4 <sup>a</sup>	148.1 ± 3.8 <sup>a</sup>
Potassium (mEq/L)	4.06 ± 0.17 <sup>a</sup>	4.30 ± 0.35 <sup>a</sup>	4.08 ± 0.44 <sup>a</sup>	3.96 ± 0.13 <sup>a</sup>
Magnesium (mg/dL)	3.97 ± 0.23 <sup>a</sup>	4.74 ± 0.32 <sup>a</sup>	3.92 ± 0.34 <sup>a</sup>	4.86 ± 0.37 <sup>a</sup>
Chloride (mEq/L)	119.0 ± 3.6 <sup>a</sup>	110.8 ± 6.9 <sup>a</sup>	110.6 ± 2.7 <sup>a</sup>	113.1 ± 4.9 <sup>a</sup>
Calcium (mg/dL)	6.28 ± 0.32 <sup>a</sup>	6.64 ± 0.25 <sup>a</sup>	7.32 ± 0.45 <sup>a</sup>	6.04 ± 0.72 <sup>a</sup>
Phosphorus (mg/dL)	6.34 ± 0.41 <sup>a</sup>	6.97 ± 1.20 <sup>a</sup>	7.35 ± 0.20 <sup>a</sup>	6.66 ± 0.19 <sup>a</sup>

**Table 2.** Serum biochemical parameters in juvenile female green sea turtles with and without fibropapillomatosis (FP). Individuals were collected in coastal waters of the southern Atlantic Ocean (Ubatuba, southeastern Brazil) from January 2011 to March 2012. They were grouped according to the severity of tumors. Data are expressed as X±SE, with sample size in parentheses. Same letters indicate mean values are not significantly different (p<0.05).

Parameter	Tumor Score			
	0 (n = 14)	1 (n = 5)	2 (n = 5)	3 (n = 7)
AST (U/L)	110.4 ± 14.8 <sup>a</sup>	90.6 ± 19.8 <sup>a</sup>	74.4 ± 24.6 <sup>a</sup>	96.7 ± 22.7 <sup>a</sup>
ALT (U/L)	14.4 ± 1.0 <sup>a</sup>	12.6 ± 1.0 <sup>a</sup>	12.8 ± 0.37 <sup>a</sup>	16.0 ± 2.1 <sup>a</sup>
ALP (U/L)	21.9 ± 4.6 <sup>a</sup>	12.0 ± 3.0 <sup>a</sup>	16.0 ± 8.0 <sup>a</sup>	16.4 ± 4.0 <sup>a</sup>
GGT (U/L)	4.71 ± 1.66 <sup>a</sup>	1.00 ± 0.32 <sup>a</sup>	1.20 ± 0.49 <sup>a</sup>	2.3 ± 1.1 <sup>a</sup>
CK (U/L)	1598.4 ± 496.7 <sup>a</sup>	468.4 ± 149.0 <sup>a</sup>	859.2 ± 257.6 <sup>a</sup>	776.0 ± 257.6 <sup>a</sup>
LDH (U/L)	396.8 ± 127.6 <sup>a</sup>	150.2 ± 43.3 <sup>a</sup>	228.6 ± 70.9 <sup>a</sup>	174.6 ± 66.0 <sup>a</sup>
HMGR (U/mg protein)	1.63 ± 0.20 <sup>a</sup>	0.70 ± 0.26 <sup>b</sup>	0.73 ± 0.10 <sup>b</sup>	0.89 ± 0.16 <sup>b</sup>

**Table 3.** Serum enzyme activity in juvenile female green sea turtles with and without fibropapillomatosis (FP). Individuals were collected in coastal waters of the southern Atlantic Ocean (Ubatuba, southeastern Brazil) from January 2011 to March 2012. They were grouped according to the severity of tumors. Data are expressed as X±SE, with sample size in parentheses. Same letters indicate mean values are not significantly different (p<0.05). AST: aspartate aminotransferase; ALT: alanine aminotransferase; ALP: alkaline phosphatase; GGT: gamma glutamyl transferase; CK: creatine kinase; LDH: lactate dehydrogenase; HMGR: 3-hydroxy-3-methylglutaryl-CoA reductase.

$\pm 5.7$  cm), 5 green turtles were moderately afflicted with FP (score 2; CCL =  $37.6 \pm 2.1$  cm), and 7 green turtles were heavily afflicted with FP (score 3; CCL =  $44.4 \pm 4.2$  cm). There were no significant differences in CCL among these groups of green turtles.

No significant difference was observed in hematological parameters among the four groups of green turtles analyzed (Table 1). However, green turtles moderately or heavily afflicted with FP showed significantly lower serum cholesterol concentration than those non-afflicted or lightly afflicted with FP (Table 2). Also, green turtles with FP (lightly, moderately and heavily afflicted with FP) had significantly reduced serum HMGR activity respect with those non-afflicted with FP (Table 3). Indeed, a significant and positive correlation was observed between serum HMGR activity and cholesterol concentration ( $r = 0.48$ ;  $p = 0.01$ ).

Analysis of blood parameters is a useful tool to evaluate the health condition of turtles, as they can provide information for the diagnosis and prognosis of diseases. Furthermore, they have been used as indicators of physiological changes due to illness, stress or exposure to environmental contaminants (Omonona *et al.* 2011). Therefore, hematological and blood chemistry analyses can be considered important steps in determining the physiological and pathological conditions in turtles (Gelli *et al.* 2009).

Among all the reported effects, it is worth noting that only 2 out of the 34 parameters analyzed in the present study significantly varied among the groups of green turtles with different tumor scores. In this case, reduced serum HGMR activity and blood serum cholesterol concentration were observed in the green turtles heavily (score 3) or moderately (score 2) afflicted with FP compared with those lightly (score 1) or non-afflicted (score 0) with FP. Furthermore, serum HMGR activity was also lower in green turtles lightly afflicted with FP than in those non-afflicted with FP.

According to Aguirre & Balazs (2000), turtles less than 35 cm are immature and those with CCL within the range of 35 to 65cm are juveniles. This scheme for grouping sea turtles was also adopted by Labrada-Martagón (2010). Therefore, according to this scheme, immature and juvenile female green turtles were evaluated in the present study. Also, based on CCL range, green turtles analyzed in the present study could be considered as post-pelagic juveniles (Santos *et al.* 2015). In this context, it is worth noting that no significant difference in CCL was observed among the four groups of green turtles analyzed in the present study. Therefore, conditions described above may have minimized the potential high variability in the response of biochemical and physiological parameters which would be associated with intrinsic factors, such as sex and body size (Aguirre *et al.* 1995; Camacho *et al.* 2013). Indeed, they could help to explain the discrepancy among our findings and those from previous studies with green turtles with and without FP (Foley *et al.* 2005; Aguirre *et al.* 1995; Work & Balazs 1999; Aguirre & Balazs 2000; Santos *et al.* 2015).

It is important to note that reduced serum cholesterol concentration, as observed in the present study, was also reported for green turtles with FP from other regions (Aguirre *et al.* 1995; Aguirre & Balazs 2000; Work *et al.* 2001, 2003). This finding suggests that the observed drop in serum cholesterol concentration may be a primary response of green turtles to FP. In turn, the reduced serum cholesterol concentration may be related to the reduced activity of serum HMGR observed in green turtles

afflicted with FP. Indeed, the level of reduction in serum cholesterol concentration (53.6%) was paralleled by a quite similar reduction (48.5%) in serum HMGR activity. Furthermore, a significant and positive correlation was observed between these two parameters in this study. It is worth noting that HMGR catalyzes the four-electron reduction of 3-hydroxy-3-methylglutaryl-CoA (HMG-CoA) to coenzyme A (CoA) and mevalonate, which is the rate-limiting step in sterol biosynthesis (Kritchevsky & Kritchevsky 1992; Holdgate *et al.* 2003). Therefore, an inhibition of serum HMGR activity would induce a lower rate of cholesterol synthesis, thus leading to a reduced level of serum cholesterol, as observed in green turtles with FP evaluated in the present study.

In addition to the influence of the reduced HMGR activity, as discussed above, a higher rate of cholesterol oxidation could also help to explain the lower concentration of serum cholesterol observed in green turtles moderately and heavily afflicted with FP compared with those non-afflicted or lightly afflicted with FP. Some possible causes of a higher rate of cholesterol oxidation would be an excessive exposure to ultraviolet radiation (Morin *et al.* 1991) and/or the oxidative stress induced by exposure to environmental contaminants (Monserrat *et al.* 2007). Although the current thinking is that FP is associated with a herpesvirus infection (Rodenbusch *et al.* 2012, 2014), these environmental factors may trigger processes that influence FP expression and lesions (Aguirre *et al.* 1994; Santos *et al.* 2010; Van Houtan *et al.* 2014). Therefore, future studies should address the influence of UV and aquatic contaminants on the rate of cholesterol oxidation in green turtles for a better understanding of FP etiology. In fact, abnormally low concentration of serum cholesterol is reported as being a reliable biomarker of malignancy in humans (Ahn *et al.* 2009).

In summary, data reported in the present study indicate that reduced serum HMGR activity and cholesterol concentration are adequate and reliable biomarkers of FP in immature and juvenile female green turtles. Further studies are needed to determine whether these biomarkers may also be applied to juvenile male green turtles.

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## Dog Attacks on Loggerhead Turtles Nesting in Greece

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Nesting sea turtles may suffer attacks from various land predators such as jackals (Peters *et al.* 1994), jaguars (Troëng 2000, Arroyo-Arce & Salom-Pérez 2015, Alfaro *et al.* 2016), coyotes (Drake *et al.* 2003), and dogs (Caldwell 1959, Santos & Godfrey 2001). As far as can be ascertained, there has never been a documented dog attack on a nesting sea turtle in the Mediterranean. The Greek NGO ARCHELON has conducted morning and night surveys on nesting beaches of Greece since the beginning of 1980s. Although stray dogs and foxes regularly visit nesting beaches to predate on turtle eggs or hatchlings, we had never recorded a dog attack on a nesting turtle, until recently in Kyparissia Bay, Greece.

During the nesting seasons of 2014 and 2015, adult female loggerhead turtles were found severely injured at the nesting area of southern Kyparissia Bay, western Peloponnese. In recent years, this 9.5km nesting beach hosts what is considered to be the largest nesting loggerhead aggregation in the Mediterranean (Margaritoulis *et al.* 2015).

The injured turtles were encountered mostly during the night surveys, when individual nesting females are tagged and measured. Injured turtles bore severe wounds on both front limbs at the shoulder area. The skin in this area was torn off and the muscles eaten by the dogs, exposing the bones. In one case, an injured turtle was found on the beach, during a morning survey, unable to move due to its severe injuries (Fig. 1). Identification of dogs

as the attacking animals was initially deduced from their tracks in the sand. Subsequently, during a night survey a pack of three stray dogs was directly observed attacking a nesting turtle. It should be noted that no golden jackals (*Canis aureus*), a known predator of adult turtles in the Mediterranean (Peters *et al.* 1994), exist in this area (Giannatos *et al.* 2005).

Examination of dog tracks and blood stains on sand along the turtles' crawls indicated that the attacks occurred mostly during the procedure of digging or egg-laying. In 2014, 12 individual turtles were found injured by dogs, seven of which were transported to ARCHELON's Rescue Centre (RC) in Glyfada (Fig. 2), three turtles were treated locally and released, and the remaining two turtles died on site before transport to the RC. All seven turtles admitted to the RC were eventually released following varying rehabilitation durations. The total number of injured turtles was certainly more than the 12 found, as blood stains were observed on the sand along several other turtle nesting crawls, as well as during night surveys several turtles were observed bearing partly healed bite marks at the same locations as those attributed to dog attacks.

To counter the threat to nesting turtles from the stray dogs, special night patrols were organized to chase the dogs off the beach. Further, known dog owners near the beach were visited to request that they keep their dogs restrained at night, which all owners reported doing. Moreover, an attempt to catch the stray dogs with



**Figure 1.** Emergent adult female turtle unable to move due to severe injuries on fore limbs caused by stray dogs on 25 June 2014 in Kyparissia Bay. The turtle did not nest and died on site, before transportation to ARCHELON's Rescue Centre.





**Figure 2.** Loggerhead turtle, injured by dogs while nesting on 30 June 2014 in Kyparissia Bay, is prepared for transportation to ARCHELON's Rescue Centre.

live traps failed. Dog attacks to nesting turtles finally ceased in late July 2014, before the end of the nesting period, suggesting that the stray dogs had left the area.

During the subsequent nesting season (2015) there were also similar attacks, albeit to a lesser extent: one turtle was transported to RC while five other females were treated locally and released after a few days. The reduced number of attacks in 2015 was possibly a result of running special night-patrols from the beginning of the nesting season (2 June) to discourage dogs from attacking nesting turtles. No other incidents were recorded in subsequent seasons, including 2018. Therefore, we conclude that the attacks were inflicted by an occasional group of stray dogs.

Continuous removal of reproductive females may have a severe impact on a sea turtle population (Margaritoulis & Touliaou 2011). However, the relatively low number of attacks compared to the large number of nesting females in this area (>1200 nests/yr in both 2014 and 2015 with no subsequent decline in numbers), as well as the eventual cessation of attacks, indicates that the overall impact of these attacks to the loggerhead population in southern Kyparissia Bay was minimal.

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# Sea Turtle Records at the Environmental Protection Area of Algodual-Maiandeuá, Para State, Brazil

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The Para State Coast, in Northern Brazil, is well known for its dynamic environment and high primary productivity as the Amazon and Tocantins Rivers meet the Atlantic Ocean (de Matos & Lucena 2006). Despite fishermen reports of sea turtle occurrence along the coast (Brito *et al.* 2015), there is a lack of documentation and publications regarding sea turtles in the area. The same dynamic features that makes this region unique, also present a challenge for access to remote regions of the littoral zone.

Previous telemetry studies have reported the use of Para state coast as a transit and forage area by post-nesting green sea turtles, *Chelonia mydas* (Baudouin *et al.* 2015; Chambault *et al.* 2015), loggerhead turtles, *Caretta caretta* (Marcovaldi *et al.* 2010), hawksbill-loggerhead hybrids, *Eretmochelys imbricata* and *Caretta caretta* (Marcovaldi *et al.* 2012), and olive ridley turtles, *Lepidochelys olivacea* (Silva *et al.* 2011). All these observations highlight the importance of sea turtle monitoring efforts in the area.

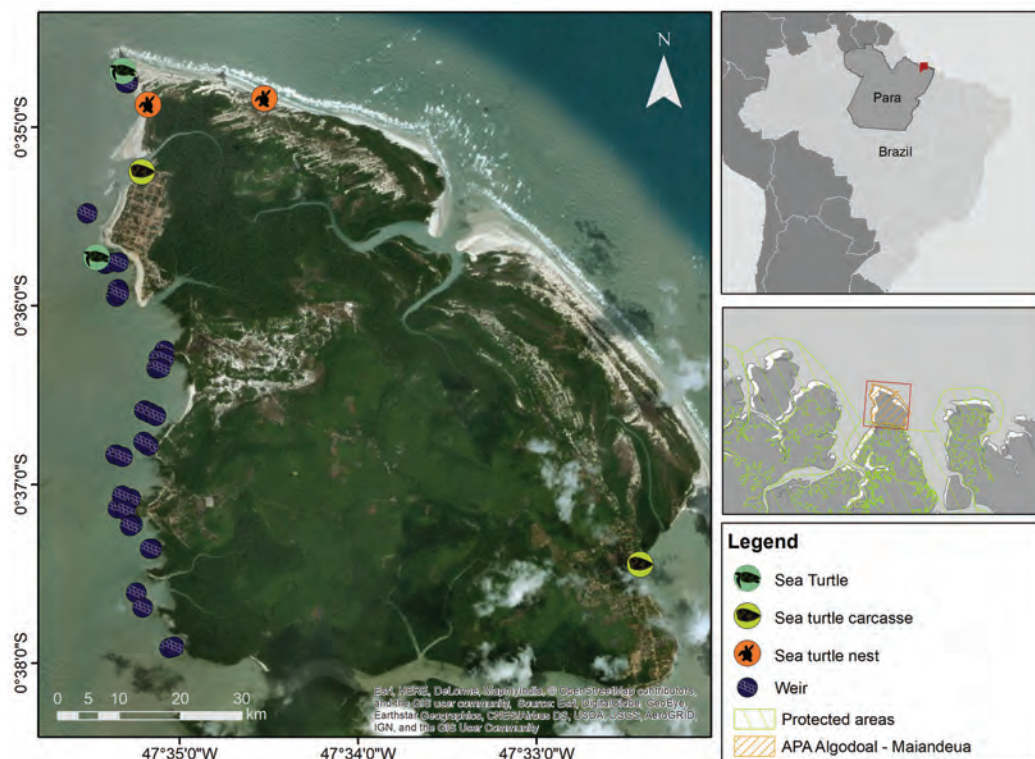
Here we report sea turtle data collected over two years of sporadic monitoring of the Environmental Protection Area of Algodual-Maiandeuá (APA Algodual-Maiandeuá), located in the Maracana municipality. We visited the island on five occasions between 2013

and 2014, collecting data on nesting activity, bycatch and recording sea turtle carcasses (Table 1, Fig. 1). All data were first reported by fishermen and confirmed by us *in situ*.

We received the first call reporting a nesting activity on 16 March 2013. We monitored the nest on four occasions: recently after egg laying; mid-development; around the predicted day of emergence; and the day of emergence (Fig. 2). After emergence, we confirmed that the nest was laid by a hawksbill turtle, with a total of 135 live hatchlings, four dead in the nest and 35 unhatched eggs, among them 10 eggs with fungus.

We also observed two live juvenile green sea turtles during the second and third trips to monitor the hawksbill nest. The turtles were caught in different fishing weirs (Fig. 3), and brought to shore, where we collected morphometric data (Table 1, Fig. 3). The turtles were released immediately after the data collection, into areas adjacent to where they were caught.

During the last trip in June 2014, we also visited the village of Fortalezinha, located on the southeast portion of the Island. On this visit we documented and measured five preserved sea turtle carapaces. They were reported as having stranded on the shore



**Figure 1.** Sea turtle areas of occurrence within the limits of the APA Algodual-Maiandeuá. Inset maps show the overall location of the protected area in relation to Para State Coast.

Code	Date	General location	Lat.	Long.	Sp.	CCL	CCW	State	Development
ALG15MAR13-01	15-Mar-13	Princesa	-0.581	-47.575	Ei	NA	NA	alive	Adult/nest
ALG13ABR13-01	13-Apr-13	Weir	-0.578	-47.589	Cm	34	30	alive	Juvenile
ALG05MAY13-01	5-May-13	Weir	-0.595	-47.591	Cm	33.5	31	alive	Juvenile
ALG05JUN14-01	5-Jun-14	Fortalezinha	-0.624	-47.540	Cm	46.7	43.6	shell	Juvenile
ALG05JUN14-02	5-Jun-14	Fortalezinha	-0.624	-47.540	Cm	32.5	29.5	shell	Juvenile
ALG05JUN14-03	5-Jun-14	Fortalezinha	-0.624	-47.540	Cm	34.2	30	shell	Juvenile
ALG05JUN14-04	5-Jun-14	Fortalezinha	-0.624	-47.540	Cm	37.7	33.3	shell	Juvenile
ALG05JUN14-05	5-Jun-14	Fortalezinha	-0.624	-47.540	Cm	37	33.9	shell	Juvenile
ALG06JUN14-01	6-Jun-14	Furo Velho	-0.588	-47.587	Cm	NA	NA	head	Adult or subadult

**Table 1.** Sea turtle records at APA Algodual-Maiandeuá. Sp. = Species, Ei = *Eretmochelys imbricata*, Cm = *Chelonia mydas*, CCL= curved carapace length (cm), CCW= curved carapace width (cm).



**Figure 2.** Princesa Beach hawksbill (*Eretmochelys imbricata*) nest monitoring stages. a) nest monitored 2 days after deposition. b) full view of the nest location. c) mid-nest development monitoring. d-f) day of emergence.



**Figure 3.** Juvenile green sea turtle (*Chelonia mydas*) caught in local weirs during the 3<sup>rd</sup> and 4<sup>th</sup> visits to monitor the hawksbill nest. a-c) Juvenile sea turtle captured on 13 April 2013. d-f) Juvenile sea turtle captured on 05 May 2013.

**a**

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**Figure 4.** Green sea turtle (*Chelonia mydas*) remains from two different locations on the island. a-e) Juvenile green sea turtle carapaces. f) Subadult or adult green sea turtle head with preserved skin tissue.

in the surrounding area, however the exact date they were found remains unknown (Fig. 4a-e). Other evidence of green sea turtle remains was reported by a local fisherman during our last visit to the island in 2014. He reported that a carcass was found near the Furo Velho tidal channel (Fig. 1). However, he only saved the head, which based on its size, we estimated to come from a subadult or adult green sea turtle (Fig. 4f).

The present note provides evidence of sea turtle presence in APA Algodal-Maiandua in Para state. It is important to recognize that this monitoring effort was entirely self-funded, which resulted in punctuated and non-standardized sampling occasions. We would like to raise awareness regarding the presence of sea turtles within the APA Algodal-Maiandua protected area, and urge that studies be carried out to understand the complexity of use of the area by sea turtles of distinct species and life stages. Improving research and monitoring will allow a better accounting of the uses of this protected area for nesting and foraging and inform conservation efforts within the region.

# Strandings of Olive Ridley Sea Turtle, *Lepidochelys olivacea* Eschscholtz, 1829 from the Coastal Waters of the United Arab Emirates

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The olive ridley sea turtle, *Lepidochelys olivacea* Eschscholtz, 1829 is one of seven extant species of marine turtles and one of two species of the *Lepidochelys* genus (Limpus 2008; Manire *et al.* 2017). This genus consists of the smallest marine turtle species and its members are unique in having a pore on each of the four pairs of inframarginal scutes (Marcovaldi 2001). The anatomical differences between *Lepidochelys olivacea* and *Lepidochelys kempii* include differences in jaw morphology and head size; the olive ridley having the smaller head and its carapace having 6-10 pairs of lateral scutes (Marcovaldi 2001; Marquez 1990; Manire *et al.* 2017). The Kemp's ridley geographic distribution is limited to the Gulf of Mexico and the western Atlantic basin while olive ridleys occur all through the tropical oceans (Pritchard 1969; Manire *et al.* 2017).

Olive ridley sea turtles have been reported to occur in both neritic and oceanic zones. Excluding the Gulf of Mexico, they nest throughout tropical waters in approximately 60 countries, however, their migrations are less understood (Pritchard 1969; Bowen *et al.* 1997; www.redlist.org). Their migratory routes comprise tropical and subtropical zones: northwest, eastern central, southwest, western central Pacific Ocean; northeast, northwest, eastern central, southeast, southwest, western central Atlantic Ocean; and eastern, western Indian Ocean (Pritchard 1969; Abreu-Grobois & Plotkin 2008; Manire *et al.* 2017).

In the Western Indian Ocean, olive ridleys have been observed in the waters of Mozambique, Somalia, Madagascar, South Africa, Kenya, Maldives, Pakistan, India, Iran and Oman (Abreu-Grobois & Plotkin 2008). In Oman, olive ridleys are known to nest at Masirah Island (Rees *et al.* 2012). In the Arabian Gulf, olive ridley turtles were first recorded in 2003 by Bishop *et al.* (2007) in the coastal waters of Kuwait, and has since been observed in the coastal waters of Iran (Qeshm Island, Larak Island, Bushehr town, Kharg Island) and Bahrain (Abdulqaader & Miller 2012; Rees *et al.* 2012; Tollab *et al.* 2015). Furthermore, in May 2013, an olive ridley female was recorded nesting for the first time in the Arabian Gulf at Nayband Marine-Coastal National Park, Iran (Tollab *et al.* 2015).

Published records of olive ridley sea turtles in the coastal waters of the United Arab Emirates are rare on both the western (Arabian Gulf) and eastern (Gulf of Oman) coasts. In the Gulf of Oman, satellite tracking has revealed that, during their northern post nesting migrations from Masirah Island, some olive ridleys will settle for lengthy periods in waters of Pakistan, Iran and the eastern coast of the United Arab Emirates (Rees *et al.* 2012). There is one stranding record from the Arabian Gulf coast of Dubai involving the rescue, rehabilitation and release of an olive ridley by the Dubai Turtle Rehabilitation Project (www.jumeirah.com). Additionally, there were two observations from the

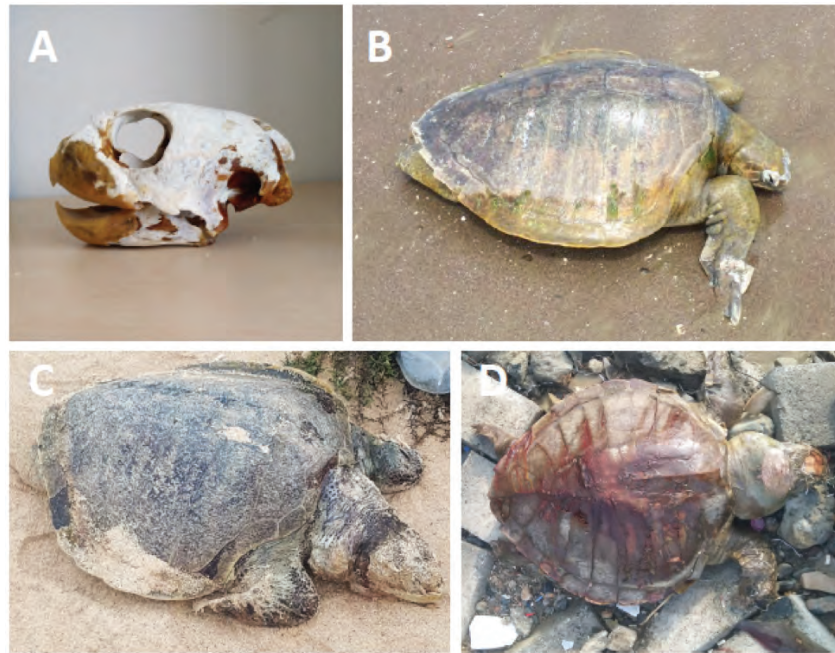
Arabian Gulf Coast of Abu Dhabi (EAD 2016), one unpublished stranding observation from the Arabian Gulf coast of Ras Al Khaima (J. Judas, personal communication, 21 May 2018) and one unpublished live observation from the Gulf of Oman coast of Fujairah (www.youtube.com/watch?v=89klrwJxJJ0). Here we present four additional records of olive ridley sea turtles from the eastern and western coasts of the United Arab Emirates.

In 2012, marine turtle skeletal remains were discovered on the beach of the Alqurm Wa Lehfhaiiah Protected Area (25.000297 °N; 56.370867 °E) in the city of Kalba, Emirate of Sharjah (J. Pereira, personal communication, 4 April 2019). The skull of that turtle was collected and stored with the EPAA research department (Fig. 1A). The turtle was identified as an olive ridley sea turtle due to the morphology of its skull and beak (rhamphotheca). The skull is triangular shaped with deep parietal notches. The beak (both upper and lower) is pointed with the upper having a wide plate and sharp edged alveolar surface and the lower has a sharp wide ridge alongside the buccal margin (Wyneken 2001).

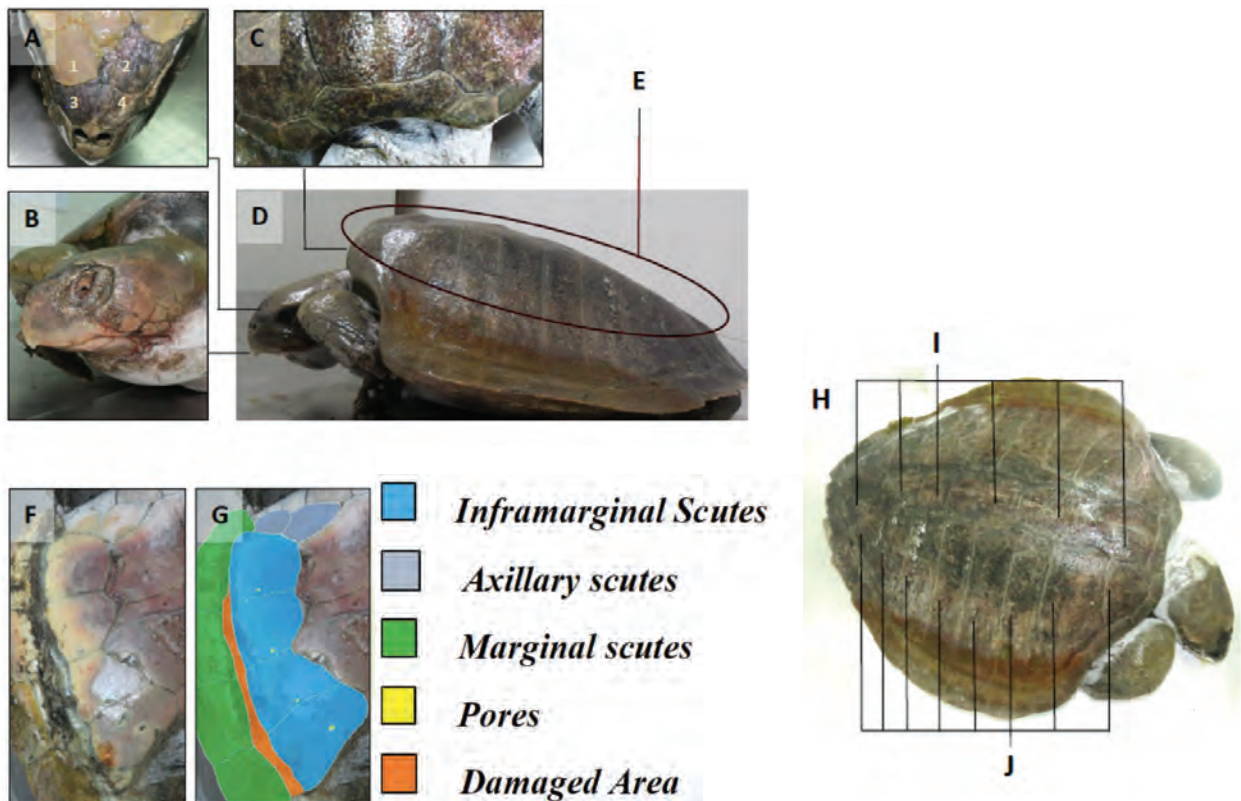
The second specimen was recorded on 13 August 2017 on the beach of Alqurm Wa Lehfhaiiah Protected Area (Fig. 1B). The specimen was observed to be a condition code 2 (Wyneken 2001; Poppi & Marchiori 2012). It had a minimum curved carapace length (CCL) of 58.7 cm and its morphology was consistent with that of an olive ridley sea turtle: 4 prefrontal scutes (Fig. 2A), a horny beak (Fig. 2B), one precentral scute that touches the anterior central scute and the two anterior lateral scutes (Fig. 2C), carapace with scutes that do not overlap (Figs. 2D, 2E), four inframarginal scutes with pores (Figs. 2F, 2G), a semi-circular carapace (Fig. 2H), six central scutes (Fig. 2I), and eight lateral scutes (Fig. 2J). The specimen was observed to have some damage to the left posterior margins of the carapace consistent with a boat strike. Damage was also observed at the right lateral area of the inframarginal scutes.

The third specimen was discovered on 25 March 2019. A stranded sea turtle was discovered on the beach of Sir Bu Na'ir Island (GPS: 25. 226239 °N, 54.218424 °E) in the Arabian Gulf coast of the UAE (Fig. 1C). The specimen was observed to be a condition code 3. Its CCL was 54.6 cm and its morphology was consistent with that of an olive ridley sea turtle. This specimen had no external signs of harmful human interactions. Finally, on 5 August 2019 a stranded olive ridley sea turtle was discovered on the coast of the city of Kalba (GPS: 25.0799223 °N, 56.3606723 °E). The specimen was a condition code 3 and had no external signs of harmful human interactions (Fig. 1B). Its CCL was 60.5 cm.

Marine migrants have essential roles in their ecosystems and many of them are at risk of the impacts of anthropogenic threats (Plotkin 2007). The observation of evidence of harmful human interaction with one of these specimens is unfortunately not



**Figure 1.** Olive ridley sea turtle strandings from the Sharjah emirate. A: Skull recovered from the beach of the Alqurm Wa Lehhafaiiah Protected Area in 2012. B: Stranding discovered on the beach of the Alqurm Wa Lehhafaiiah Protected Area on 13 August 2017 (Photo: Ahmed Al Mazmi 2017). C: Stranding discovered on the beach of Sir Bu Na'ir Island on 25 March 2019 (photo: Mohammed Saif 2019). D: Stranding discovered on the beach of Kalba on 5 August 2019.



**Figure 2.** Olive ridley sea turtle stranding species diagnosis. A: Four prefrontal scutes. B: Horny beak. C: Precentral scute that touches the anterior central scute and both anterior lateral scutes. D: lateral side of specimen. E: Scutes not overlapping. F: An unedited photo of the inframarginal scutes on the ventral lateral region of the sea turtle. G: A highlighted copy of F showing marginal scutes, axillary scutes, four inframarginal scutes with pores, and damaged areas of the plastron. H: Top view of the specimen showing an almost circular carapace. I: Six central scutes. J: Eight lateral scutes.

surprising as previous studies suggest that marine turtles in the United Arab Emirates are interacting at an increasing frequency with a variety of anthropogenic threats (Fowler *et al.* 2007; EAD 2016; Farkas *et al.* 2017; Sinaei & Bolouki 2017; Yaghmour *et al.* 2018a,b). Despite being the most numerous sea turtle species, olive ridleys are categorized as “Vulnerable” by the IUCN (www.redlist.org) and are protected throughout its range under CITES, Appendix I (Manire *et al.* 2017). For effective conservation outcomes to be achieved, sufficient knowledge of their spatial and temporal ecology is needed (Colman *et al.* 2014). This paper contributes to documenting the occurrence of this species in the United Arab Emirates, where information on this species is scarce.

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## REPORTS

### RETOMALA: 25th Annual Meeting of Latin American Sea Turtle Specialists, Charleston, South Carolina, USA - 04 February 2019

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As part of the 39<sup>th</sup> International Sea Turtle Symposium in Charleston, South Carolina, USA, the 25<sup>th</sup> Latin American Sea Turtle Specialist Meeting, RETOMALA (in Spanish: *Reunión de Especialistas Latinoamericanos en Tortugas Marinas*), was held on 04 February 2019. Annually, this regional meeting congregates researchers, local leaders, volunteers, community members, and marine turtle advocates who share their findings and experiences in a friendly environment with oral presentations and questions/answers sessions. In this year's meeting, there were registered 64 attendees from 19 different countries of Latin America and other regions.

The 25th RETOMALA's aim was to update our status of knowledge about both ridley turtles (*Lepidochelys kempii* and *L. olivacea*) in the Latin American region. Our agenda included 19 talks: one regarding regional status, 13 with national or local level data (Fig. 1), and two regarding regional initiatives (Table 1).

*Lepidochelys* turtles have long generated worldwide interest, due to their unique nesting behaviour commonly known as “*arribadas*,” a Spanish term which is defined as “the arrival on land of something that was in the sea” (Real Academia Española, www.rae.es). More recently, our interest in these species has been further raised after inopportune tragic events that have impacted both species: viral videos on social media about the impact of plastic on nesting females of *L. olivacea* in the Pacific Ocean of Costa Rica (Robinson & Figgenger 2015; Robinson *et al.* 2016), and the 2010 Deepwater Horizon oil spill that occurred in the Gulf of Mexico, the primary habitat of *L. kempii* (Putman *et al.* 2015; Reich *et al.* 2017).

According to the IUCN's Red List, *L. kempii* is categorised as critically endangered and *L. olivacea* is listed as vulnerable (www.redlist.org). Both species face multiple threats which include habitat loss, by-catch, directed take of individuals for their meat, and harvest of eggs (Valverde & Holzward 2017). These two latter threats mainly affect *L. olivacea* populations worldwide. During our meeting, presenters and attendees recognised the necessity of updates in both conservation status and available information of both Ridley's turtles in the region.

Our meeting started with a short welcome speech from the organisers. Then, the presentation of the “Sea Turtle Male Initiative” presented by Marco Garcia-Cruz, who stressed the importance of RETOMALA as a forum to investigate and document roles and

habitats male marine turtles in the wider Latin American region. Garcia-Cruz also invited our attendees to be part of this worldwide initiative (García-Cruz *et al.* 2018).

Following this, presentations focused on ridley turtles were given (Table 1). At the end of our meeting, Brad Nahill showed updated results of the campaign “Too Rare to Wear.” The presentation included educational posters, social media impact, and experiences of sharing with tourism providers and services (*e.g.*, AirBnB).

This meeting was the final chapter in the RETOMALA initiative to focus on the current status of particular species in the region. The initiative began in 2012 with a focus on *Chelonia mydas* (Barreto-Sanchez & Barrios-Garrido 2012); in 2013 the focus was *Caretta caretta*; in 2016 the focus was *Dermochelys coriacea*; in 2018 the focus was *Eretmochelys imbricata* (Barrios-Garrido *et al.* 2018). We recommend the reassessment of the status of all regional species starting in 5-10 years from now, to facilitate a better understanding of status changes over time.

One innovation from this year's meeting was live-streaming the presentations through our Facebook page (www.facebook.com/tortugeroslatinos), which allowed us to increase the participation (up to 40 online viewers) from members who could not attend in person but actively participated by direct messages during the questions/answer sessions. Currently, the videos of our meeting have reached more than 2,000 views and they are still available to watch in our web page.

#### Descriptive summary of presentations in alphabetical order by species:

*Lepidochelys spp.* Hector Barrios-Garrido presented the Conservation Enforcement Capacity index (CECi) based on both species (Barrios-Garrido 2018). This index may be used to predict the conservation status of the marine turtle species based on socio-economic indicators that may influence the conservation and enforcement capacity of national governments to protect endangered species. Three Regional Management Units (RMUs) were evaluated for this presentation, two for *L. olivacea* (Eastern Pacific and Western Atlantic) and one for *L. kempii* (North Western Atlantic and Gulf of Mexico). Based on CECi, the East Pacific RMU may be considered threatened in the future using CECi, and the other two RMUs evaluated are likely to be classified as Least Concern in future evaluations by

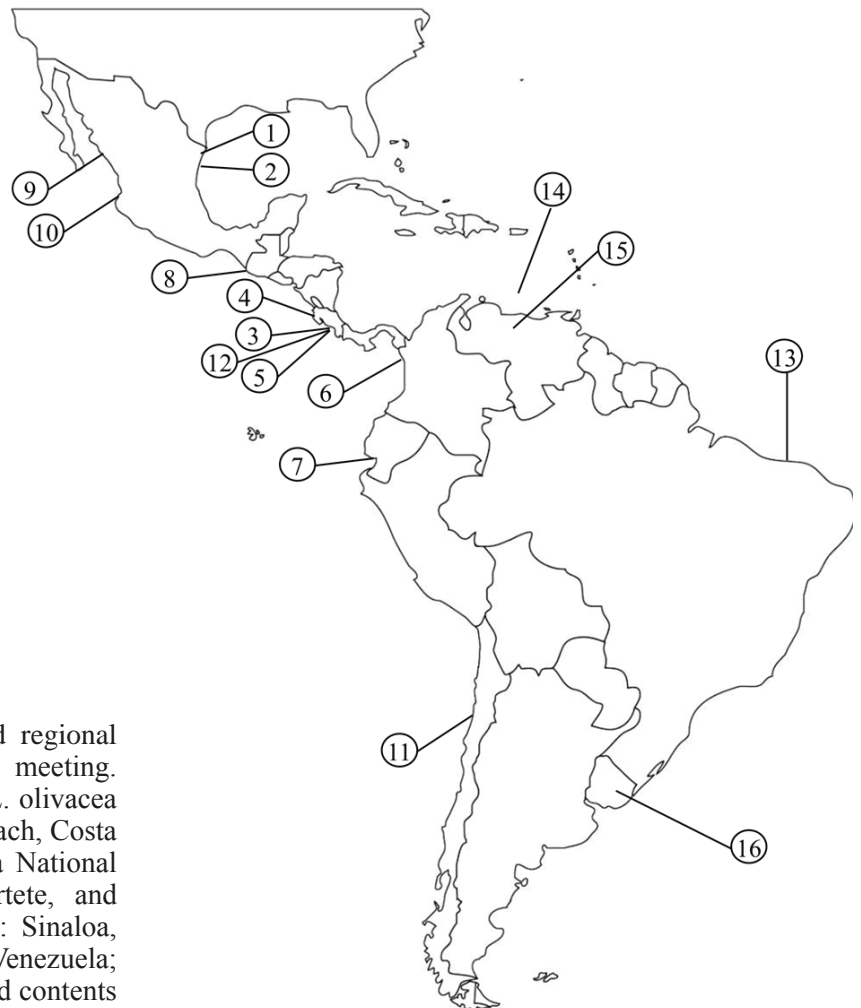


Title in English	Original title (language)	Scale of project	Presenters	Authors
Welcome to 25th RETOMALA meeting, including aim and dynamic of the meeting			J.M. Rguez-Barón & H. Barrios-Garrido	J.M. Rguez-Barón & H. Barrios-Garrido
The Global Male Sea Turtle Initiative: adding males to the conservation equation	<i>Iniciativa Global de Tortugas Macho: sumando los machos a la ecuación de conservación</i> (S)	Global	M. García-Cruz	M. García-Cruz
Conservation Enforcement Capacity index: olive and Kemp ridley's turtles in Latin America	<i>Índice de Capacidad para la Aplicación de la Conservación: caso tortuga lora y golfina en Latino América</i> (S)	Latin America	H. Barrios-Garrido	H. Barrios-Garrido & M. Hamann
<i>Lepidochelys kempii</i>				
DNA sequences of the COI gene in <i>Lepidochelys kempii</i>	<i>Secuencias del AND-gen COI en Lepidochelys kempii</i> (S)	México	M.A. Reyes-Lopez	M.A. Reyes-Lopez
Recovery of the marine turtle, <i>Lepidochelys kempii</i> in Tecolutla, Veracruz, Mexico	<i>Proceso de recuperación de la Tortuga marina, Lepidochelys kempii en Tecolutla, Veracruz, México</i> (S)	México	M. F. Manzano	M. F. Manzanon & I.E. Galván T.
<i>Lepidochelys olivacea</i>				
Egg harvesting as conservation tool of olive ridleys at Ostional beach, Costa Rica	<i>La cosecha de huevos como herramienta de conservación de la tortuga olivácea en playa Ostional, Costa Rica</i> (S)	Costa Rica	R. Valverde	R. Valverde, C.M. Orrego & L.G. Fonseca
Conservation of olive ridley turtles on nesting beaches in Nicoya Sur Peninsula, Costa Rica	<i>Conservación de tortuga lora en las playas de anidación de la Península de Nicoya Sur, Costa Rica</i> (S)	Costa Rica	C. Mejia-Balsalobre	C. Mejias-Balsalobre, D. Rojas-Cañizales, D. Arauz; I. Naranjo & R. Arauz
Arribada behaviour of olive ridley turtles at Ostional beach, Costa Rica	<i>El comportamiento de arribada en las tortugas loras en Playa Ostional, Costa Rica</i> (S)	Costa Rica	V. Bezy	V. Bezy
Ecotourism of <i>L. olivacea</i> in Colombia, with special emphasis on El Valle municipality-Bahia Solano	<i>Ecoturismo de L. olivacea en Colombia, con énfasis especial en el corregimiento de El Valle - Bahía Solano</i> (S)	Colombia	J.S. Ayala	D. Amorocho & J.S. Ayala
Current status of research and conservation of <i>L. olivacea</i> in Ecuador	<i>Situación actual de la investigación y conservación de L. olivacea en Ecuador</i> (S)	Ecuador	F. Vallejo	F. Vallejo & Equilibrio Azul
Assessment of marine turtle conservation in Guatemala	<i>Análisis situacional de la conservación de tortugas marinas en Guatemala</i> (S)	Guatemala	C. Muccio	C. Muccio & ARCAS
Population assessment and base-line study of olive ridley ( <i>Lepidochelys olivacea</i> ) health parameters at northern Sinaloa, Mexico	<i>Caracterización poblacional y establecimiento de la línea base de parámetros de salud de tortuga golfina (Lepidochelys olivacea) en el norte de Sinaloa, México</i> (S)	Mexico	A.A. Zavara-Norazaray	B.A. Espinoza-Romo, C.P. Ley-Quiñonez, J.C. Sainz-Hernandez, C. Hart, M.A. Reyes-Lopez, F.Y. Camacho-Sanchez, K.A. Zavala-Felix, V. Leal-Sepulveda & A.A. Zabala-Norzagaray
Relationship between teratogenesis, pollutants, and DNA methylation in olive ridley embryos	<i>Relación entre teratogénesis, contaminantes y metilación del ADN en embriones de tortuga golfina</i> (S)	Mexico	R. Martin del Campo	R. Martin del Campo
Stranding records of <i>L. olivacea</i> in Chile	<i>Registros de varamientos de L. olivacea en Chile</i> (S)	Chile	M. Jauregui	M. Jauregui & Qarapara NGO
New centre for sea turtle conservation at Ostional, Costa Rica	<i>Nuevo centro para la conservación de tortugas marinas en Ostional, Costa Rica</i> (S)	Costa Rica	V. Bezy	V. Bezy

**Table 1.** Program presented during 25th RETOMALA “Reunión de Especialistas Latinoamericanos en Tortugas Marinas.” Original title in Spanish (S); Portuguese (P); English (E). Continued on following page

Title in English	Original title (presentation language)	Scale of project	Presenters	Authors
<i>L. olivacea</i> in Brazil, 39 years of research and conservation	<i>L. olivacea no Brasil, 39 anos de pesquisa e conservação</i> (P)	Brazil	C.A. da Silva & B. Giffoni	J. Comin de Castilhos, C.A. da Silva, B. Giffoni, N. Marcovaldi & Projeto TAMAR
<i>L. olivacea</i> in Curaçao	<i>L. olivacea</i> in Curaçao (E)	Curaçao	A. Vreugdenhil	A. Vreugdenhil
<i>L. olivacea</i> in Venezuela: Historical records and anecdotal data	<i>L. olivacea en Venezuela: Registros históricos y datos anecdóticos</i> (S)	Venezuela	H. Barrios-Garrido	H. Barrios-Garrido, N. Wildermann, E. Debouis, M.F. González & C. Balladares
Olive ridley turtles in Uruguayan waters: the southernmost records in the South-western Atlantic	<i>Tortuga olivácea en aguas uruguayas: los registros más meridionales en el Atlántico Sur Occidental</i> (S)	Uruguay	D. Gonzales-Paredes	D. Gonzales-Paredes & Karumbé NGO
General Topics				
Too rare to wear	Too rare to wear (E)	Latin America	B. Nahill	B. Nahill
PLENARY MEETING- GENERAL OUTCOMES			All attendees	

**Table 1 continued.** Program presented during 25th RETOMALA “Reunión de Especialistas Latinoamericanos en Tortugas Marinas.” Original title in Spanish (S); Portuguese (P); English (E).



**Figure 1.** Locations of the local, national, and regional talks presented during the 25th RETOMALA meeting. Presentations were focused on *L. kempii* (1-2); *L. olivacea* (3-16). 1-2: Gulf of Mexico. 3, 5, 12: Ostional beach, Costa Rica; 4: Nicoya Peninsula, Costa Rica; 6: Utria National Natural Park, Colombia; 7: Las Palmas, Portete, and Pacoche beaches, Ecuador; 8: Guatemala; 9-10: Sinaloa, Mexico; 11: Chile; 13: Brazil; 14: Curacao; 15: Venezuela; 16: Uruguay. Details of the locations, projects, and contents of presentations in Table 1.

the MTSG-IUCN. Barrios-Garrido emphasised the importance of the implementation of this index, especially for species where the conservation status is outdated, including both ridley species.

*Lepidochelys kempii* in Mexico: genetic assessment. Miguel Angel Reyes-Lopez presented an overview about the genetic findings (COI gene - DNA sequences) in *L. kempii* from samples of nesting females in Rancho Nuevo, Mexico. Reyes-Lopez recognised the increase in nest records numbers in the area during the past 25 years. He has developed a DNA barcoding protocol, based on PCR and information analysis, and he presented a phylogenetic tree for *L. kempii* revealing an unprecedented diversity in the stock. Reyes-Lopez proposed the creation of banks in Latin America to stock DNA samples from marine turtles, to allow future analyses when more accurate and robust technologies are available.

*Lepidochelys kempii* in Mexico: population recovery. Fernando Manzano described the results from his 45 years of work on nesting *L. kempii* in Tecolutla, Veracruz, Mexico. Manzano, also known as “Papa Tortuga” (Turtle Dad), has helped to engage large numbers of young people to the call to avoid sea turtle consumption, through various social techniques, such as advertisements. Papa Tortuga and his team have achieved a major increase of the rookery in that area, from 500 nests in 1974 to 92,500 nests in 2017.

*Lepidochelys olivacea* in Brazil. Projeto TAMAR is the leading group researching sea turtles in Brazil, with 26 bases spread across the country. Through 39 years of monitoring by Projeto TAMAR, *L. olivacea* is now one of the most recovered species of sea turtle in Brazil, as demonstrated by increasing population numbers, larger ranges of nesting beaches on the Brazilian coastline, and temporal diversity with nesting individuals. During the presentation, it was noted that Brazil’s *L. olivacea* population has been increasing while Guyana and French Guiana have smaller populations and Suriname has a declining population. Historically, *L. olivacea* nesting in Brazil was found only in the state of Sergipe, but following decades of conservation actions, more turtles arrive to nest along a larger range of coastline spanning a number of northeastern Brazilian states both during and outside of the traditional nesting season. In 2006, satellite telemetry showed movement to the north and south along the Brazilian coastline but the studies in 2014 and 2016 have suggested that *L. olivacea* also cross the Atlantic to West African countries such as Mauritania and Cote d’Ivoire.

Although *L. olivacea* populations are recovering, shrimp trawling effort in the important nesting areas of Sergipe has been correlated with a negative effect on *L. olivacea* population. Long line fishing also presents threat. From 1999 to 2017, 22,000 longlines have been monitored which has shown more than 600 *L. olivacea* individuals injured mostly in the Northeast region of Brazil. Stranded *L. olivacea* observed in Brazil have been mostly females; however more recently, stranded males have been found as well. Most of these stranded turtles are found in northern Bahia and Sergipe, where recently about 50% of identified stranded individuals were male. Recently, Projeto TAMAR successfully negotiated with the Brazilian government to extend the closed season for shrimp trawling to overlap with peak nesting season for *L. olivacea*. Although these new fishing rules protect the turtles, enforcement continues to be an issue as illegal shrimp trawling occurs within the closed areas. In terms of land-based threats, fox have been documented preying on sea turtles eggs. Foxes are known to dig up nests and eat the eggs before end of incubation.

*Lepidochelys olivacea* in Chile. M. Jauregui described the relatively limited observations of stranded *L. olivacea* along the Chilean coast. The first described stranded *L. olivacea* occurred in the north of Chile in 1991. In 2009, Chile established the National Stranding Registry run by ‘Servicio Nacional de Pesca y Acuicultura (SERNAPESCA)’. The National Stranding registry contains 159 *L. olivacea* stranding reports from 2009 to 2018 (the author did not include data from the other marine turtle species), of which 119 were alive but with a low rate of successful rehabilitation (most of the live stranded turtle died after days or weeks in rehabilitation). The majority of the strandings are reported for the northern section of the country. Overall, the Chile has a much smaller number of *L. olivacea* records compared to other countries in the Eastern Pacific, likely due to cold coastal water temperatures from regional upwellings and cold sea currents. There is no necropsy protocol to process stranded turtles; moreover, the number of turtles by life stage is mostly unknown due to the advanced stage of decomposition of turtles when encountered by the stranding network.

Chile has one of the most controlled sea turtle conservation laws throughout the coast, with highly restrictive protocols for interacting with stranded turtles. Currently, only authorized government officials can interact with stranded turtles, regardless of condition. If a non-government researcher were to touch an animal, they are subject to penalties, and if the turtle were to die after, the responders could be prosecuted for the death of a protected species. This likely constrains the collection of data on stranded turtles found in Chile.

*Lepidochelys olivacea* in Colombia. Along the Colombian Pacific and inside Utria National Natural Park, the El Valle area includes La Cueva and El Almejal beaches and is considered the most important nesting rookery for *L. olivacea* in South America (Barrientos-Muñoz et al. 2014; Hineostroza & Páez 2001; Martínez & Paez 2000). The “Asociación Caguama,” a community-based group of 17 members, has been monitoring and conserving hundreds of nesting females and hatchlings every year since 2008. In 2017, members of WWF Colombia and Centro de Investigación para el Manejo Ambiental y el Desarrollo (CIMAD) trained the Asociación Caguama team on monitoring techniques for various demographic parameters of sea turtles, and introduced other useful tools for a successful and sustainable tourist project. The presenters highlighted the importance of a tourism project as a strategy for reducing local sea turtle consumption and illegal trade.

*Lepidochelys olivacea* in Costa Rica (arribadas). Roldan Valverde presented an overview about historical nesting data and trends of the olive ridley egg harvest at Ostional beach, in Costa Rica. Valverde explained multiple indices that have been used in the past to analyse olive ridleys arribadas (Simpson’s Rule) and to calculate the estimated number of effective nesting females (Gates et al. 1999). Annual numbers of harvested eggs in Ostional during the study period fluctuated between 1,500 and 8,000 nests. There was also summaries on research projects concerning arribadas at Ostional, including variation in hatchling success and its relationship with the harvesting of eggs, and assessing the effect of the egg harvest on the olive ridley nesting population at Ostional beach.

Vanessa Bezy presented a summary of her PhD dissertation concerning *L. olivacea* arribada behavior in Ostional Beach, Costa Rica. The project focused on potential mechanisms behind the synchronization of nesting that result in the arribadas. Her project was divided in three sections. The first analyzed oceanographic

and environmental parameters associated with arribadas and the numbers of turtles participating. Results showed that most arribadas occur during and after the third quarter lunar phase. In addition, the time since the last arribada and the sea level are also correlated with arribadas, while salinity, humidity and oceanic currents were correlated with the number of turtles that will come out to nest. Bezy stressed that although this information facilitates our understanding the arribada events, they are unable to correctly predict with certainty when the next arribada will happen. The second component used Unmanned Aerial Systems (UAS) to survey coastal waters near the beach during the year. She found there is a higher density of turtles closer to the shore during the rainy season (Aug-Nov), and it appears that turtles approach the inshore area a few days previous to the arribada taking place, although this was not always the case. During the 2016 November arribada, she estimated a 6 km<sup>2</sup> aggregation of *L. olivacea*, with a density of one turtle per meter squared. She also found a correlation between the density of turtles in the water and the number of turtles that come out to nest. The third component tested olfactory cues potentially used by turtles to engage in arribada behavior, which Bezy invited the audience to learn more about by seeing her poster.

*Lepidochelys olivacea* in Costa Rica (solitary nesting). Carmen Mejias-Balsalobre presented a summary of 20 years of research carried out by Crema (previously known as Pretoma) in the Nicoya Sur Peninsula, in Costa Rica. Crema monitored five nesting beaches =: San Miguel (1998-2018), Bejuco (2016-2018), Caletas (2002-2015), Costa de Oro (2012-2018), and Corozalito (2008-2018). The latter beach is normally visited by solitary nesting females and infrequently by 'small arribadas,' which were defined as arribadas with fewer nesters than Ostional beach. In summary, a positive trend has been documented at all the beaches evaluated in the region. However, further research is needed to verify this positive trend.

*Lepidochelys olivacea* in Costa Rica (conservation). Vanessa Bezy presented a summary of an initiative to establish a new center for the conservation of marine turtles in Ostional, Costa Rica. Given that it is an important area for *L. olivacea*, this group is trying to create a center that focuses on sea turtles and inspires visitors towards conservation and having a sustainable life. This initiative was established as higher numbers of tourist started to visit Nosara in recent years, with the resulting impacts including light, noise, and land waste production. Currently, Ostional has only an open-air landfill close to the Nosara River that flows out to the ocean in Ostional. The community in Nosara is committed and passionate for environmental conservation, and is concerned about future development that would make them more like to town of Tamarindo, which is known for high levels of tourism and struggles with issues associated with pollution. The group in Nosara is using sustainability as the main focus for tourism, including the social, economic, and tourism aspects. They have worked with the tourism sector to recognize and establish a group of essential stakeholders in the region. These stakeholders are in favor of the Ostional National Wildlife Refuge and support environmental protection. They also have formed a group focused on creating the center for sustainability, which will have facilities for research, and tourism management and logistics. The goal is that all activities at the center must be linked to sustainability.

*Lepidochelys olivacea* in Curaçao. Ard Vreugdenhil spoke about *L. olivacea* in Curacao, which historically has no records of this

species. In 2016, two live injured *L. olivacea* turtles were found, treated, and released. Researchers are currently waiting for the next *L. olivacea* sighting.

*Lepidochelys olivacea* in Ecuador. Felipe Vallejo presented information on *L. olivacea* in Ecuador on behalf of Equilibrio Azul, a nonprofit organization founded in 2004 that investigates, educates, and conserves the marine environment in Ecuador. Las Palmas, Portete, and Pacoche are significant nesting beaches for *L. olivacea*, and are where information on the species is collected. Portete beach is considered the most in need of conservation, when in 2008 all hatchlings produced were predated by dogs. Equilibrio Azul monitored the beach, where between 2012 and 2015 they successfully released an average of 33,100 hatchlings annually. In April of 2016, an earthquake struck the primary nesting area for *L. olivacea* nesting, impeding surveys and data collection. Starting in 2017, the Ecuadorian Ministry of the Environment took over the monitoring of nests laid at Portete, with 52 hatchlings released in 2017 and 136 hatchlings in 2018. Nest monitoring in Las Palmas, Esmeraldas began in 2017 with 213 hatchlings released and 109 hatchlings released in 2018. Las Palmas is located near a large urban area and an oil refinery. In Pacoche Wildlife Refuge, nest monitoring began in 2012 with 144 hatchlings released. In the 2014-2015 season, 191 hatchlings were successfully released, and in the 2015-2016 season, 337 hatchlings reached the ocean. At the time of this presentation, monitoring was still underway for the 2017-2018 season. Currently, the collaboration of the Ecuadorian Ministry of the Environment and NGOs to monitor and protect nests at Ecuadorian beaches results in around 1,000 hatchlings annually reaching the ocean from the monitored locations. Although *L. olivacea* nest numbers are increasing, large threats such as dog predation on hatchlings and injuries related to fishing practices continue and need addressing.

*Lepidochelys olivacea* in Guatemala. The ARCAS group presented an assessment of sea turtles in Guatemala, with an analysis of the population sustainability of the current quota of 20% of turtle eggs collected to be incubated in hatcheries and the remaining 80% used for commercial purposes. The group conducted the research by interviews, data collection, Google Earth, track counts and a population analysis. They reported that a 50 km extension of nesting beach in the Caribbean is restricted from egg collection. Additionally, they found that unlike in the past, currently there is little use of shell or meat consumption. A principal threat for turtles in the Caribbean is marine debris in the coastal area. Most of the nesting occurs in the southern Pacific coast of Guatemala and 99% of nests are laid by *L. olivacea*. Most eggs are collected by professional egg collectors, and according to law, 20% of each nest must be transported to a hatchery for protected incubation. However, most hatcheries are run locally and independently and do not follow proper data collection protocol, although some of them are well funded and managed. ARCAS covers seven nesting beaches with local researchers from July to December. Since 1997, they have been conducting track counts, standardizing the protocol in 2003. They found that the nest density has doubled in the past 14 years: in 2017 there were 28,506 nests in a 254 km span, counting for 2.6 million eggs with an estimated economic value of 2.9 million Guatemalan quetzals, which represents US\$ 395,340 and a resale value of US\$ 1.6 million (12.7 million Guatemalan quetzals). Furthermore, retrieved eggs placed in hatcheries have increased.

*Lepidochelys olivacea* in Mexico: population status. Alan Zavala-Norazagaray presented a summary of 14 years of work on sea turtles in Sinaloa, Mexico, focusing on *L. olivacea*. The researchers have analysed strandings, worked with local fishermen, and learned to do in-water captures. This area is an important feeding ground for five species of sea turtles, and the group has found *L. olivacea* is the most abundant, with records of juveniles, sub-adults, adults, and even lost-year individuals. Through in-water research, the researchers developed a body condition and health assessment for turtles through collecting blood and other samples. From the biochemistry analyses they established that the population has a 50% female sex ratio. Additionally, from the health assessment they concluded that turtles were largely healthy with a good body condition. This project established the first published biochemistry reference values for the species in a foraging area, amongst other publications related to sea turtle health (Zavala-Norazagaray et al. 2014; Espinoza-Romo et al. 2018; Mejia-Radillo et al. 2019). A highlight of this last publication is that they found pandemic strains of fibropapillomatosis in sea turtles and argue that some species could be acting as a vector for these strains during their annual migrations.

*Lepidochelys olivacea* in Mexico: genetic assessment. Martin del Campo presented a summary of his PhD dissertation project on teratogenesis in sea turtles. The primary objective of the research focused on establishing a relationship between the presence of contaminants such as organochlorines (e.g., Endosulfan) and mercury with the presence of congenic malformations. The project used *L. olivacea* embryos from nests laid in the north of Mexico (Sinaloa). Some of the findings included a positive relationship between presence of Endosulfan in embryos and congenic malformations, and its absence in normal embryos. Additionally, there was a significant difference in the mercury concentration in embryos with malformations (high concentration) vs. normal individuals (low concentration).

*Lepidochelys olivacea* in Uruguay. Karumbé (an NGO) presented a summary of the 15 *L. olivacea* reports along the coast of Uruguay from the previous 30 years (González-Paredes et al. 2017). These data came from three sources: Jack Frazer during a trip to Uruguay in 1983, an ICCAT report in 2014 and from Karumbé's database. The records came from museum specimens, stranding and bycatch records, and from observations made in bars and markets (see details in González-Paredes et al. 2017). One *L. olivacea* was a live 70 cm CCL turtle with no front flippers. In two cases, DNA samples were used to confirm species identification. These cases represent the southernmost reports of the species in the Southwestern Atlantic Ocean.

*Lepidochelys olivacea* in Venezuela. This presentation was based on data compiled by the national stranding database network, organized by the Biological Diversity National Office (coordinated by C. Balladares) and data retrieved from the Gulf of Venezuela, one of the most important feeding areas in the country (based on Barrios-Garrido 2018). The occurrence of *L. olivacea* in Venezuela is infrequent. Between 2002 and 2018, only 19 records (3% of all sea turtle records nationally) of *L. olivacea* were registered. It is the sea turtle species with the fewest records at the national level. In the Gulf of Venezuela, since 1987 only 8 records of this species have been documented, representing 0.38% of all 1,311 sea turtle stranding records on this important feeding ground. Due to its rarity, there is relevant information about bio-ecological aspects gathered

from the strandings (Delgado-Ortega et al. 2009; Wildermann & Barrios-Garrido 2012).

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# The Third Annual Workshop on the Use of UAS in Sea Turtle Research and Conservation at the 39th International Sea Turtle Symposium - 02 February 2019

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The 39<sup>th</sup> International Symposium on Sea Turtle Biology and Conservation, held 2-8 February 2019 in Charleston, South Carolina, USA, was host to the third ISTS Unmanned Aircraft Systems (UAS) workshop, entitled, “Problem Solving, Turnkey Systems, and What’s Next.” The enthusiastically received workshops at the two previous symposia in Las Vegas, Nevada and Kobe, Japan were aimed, by design, at introducing new and prospective users to UAS and the information that could be accessed by their use in sea turtle research. As the technology is maturing and becoming more accessible, organizers RRC, AFR, and TW shifted the focus of this workshop to dissemination of information on the “nuts and bolts” of UAS field operations, and optimal setups. The day-long format generously provided by the Symposium organizers allowed for a full morning of presentations and discussion, followed by an afternoon of practical demonstrations. With over 20 other workshops for Symposium attendees to choose from, the UAS workshop saw a moderate attendance of approximately 35 people participating for the entire day, and many interested drop-ins from other sessions.

After opening remarks by RRC, experienced UAS-users Vanessa Bezy, Elizabeth Whitman, Milton Levin, TW, Nerine Constant, and Katia Ballorain provided overviews of their drone-based research programs which included in-water and beach surveys for distribution and abundance, behavioral studies, habitat assessments, and non-sea turtle applications. Presenters were specifically asked to provide information on 1) the questions that they address with drones, 2) the actual drone system (*e.g.*, tools) that they utilize, 3) an overview of the project and logistics, including 4) type of data being collected, 5) problems encountered and recommendations, and 6) legal and safety aspects of UAS use.

Following the presentations, RRC moderated a lively discussion wherein many workshop participants shared their experiences and UAS application interests. The latter ranged from remote area surveys and workload reduction to poaching interdiction. The attendees, organizers, and presenters all exchanged valuable insights, ideas and problem-solving tips to facilitate their research and drone use.

The workshop reconvened after lunch and participants circulated among five demonstration stations that were spread around the room: RealFlight simulator. Allowed participants to virtually fly a variety of aerial drone platforms on a computer, controller and large-screen monitor setup.

Drone and ROV (Remotely Operated underwater Vehicle) hands-on. Several rotorcraft (DJI Inspire, Phantom 4 Pro, Mavic Pro 2), a fixed-wing (SenseFly eBee), two ROVs (Deep Trekker DTG2, Power Vision PowerRay), and field charging equipment were made available for attendees to inspect and manipulate.

Three software demonstration setups (Litchi, Pix4D, and Sentera Field Agent). These provided the opportunity for some hands-on experience with widely-used software for flight-planning and post-processing of imagery.

At the end of the workshop RRC and TW fielded final questions, gave wrap-up remarks, and sought feedback from the workshop attendees. Workshop outcomes will include a repository of information presented at the meeting and a Sea Turtle UAS listserv to be launched on Google Groups. The latter will provide a forum for users to help each other problem-solve and share equipment reviews and recommendations.



## RECENT PUBLICATIONS

This section consists of publications, books, reports, and academic theses that feature subject material relevant to marine turtles. Most references come from major search engines, and the editors encourage authors to submit their publications directly by email to the Recent Publications editor: [mntrecentpubs@gmail.com](mailto:mntrecentpubs@gmail.com).

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