

Marine Turtle Newsletter

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Olive ridley hatchling on Darcy Island, Australia - see pages 9-11 (photo: Roy Teale).

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Editorial: Plastic Pollution: An Ocean Emergency

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The oceans have become one giant refuse bin for all manner of plastics. Environmental and health concerns associated with plastic pollution are a long recognised international problem (Carpenter & Smith 1972). Whilst approximately 10% of all solid waste is plastic (Heap 2009), up to 80% of the waste that accumulates on land, shorelines, the ocean surface, or seabed is plastic (Barnes et al. 2009).

Plastics have an array of unique properties: they are inexpensive, lightweight, strong, durable, corrosion resistant, and with high thermal and electrical insulation properties. This versatility has revolutionised our life and not least made information technology and electrical goods far more readily available than would have been possible otherwise. They have also contributed to our health and safety (e.g., clean distribution of water and breakthrough medical devices), and have led to substantial energy savings in transportation. Unsurprisingly, with an ever expanding population and our standard of living continuously improving, plastic production has increased from 0.5 to 260 million tonnes per year since 1950 (Heap 2009), accounting today for approximately 8% of world oil production (Thompson et al. 2009b). Almost all aspects of our daily life involve plastics in some form or another: from hair dryers to shoes, to the car we drive and the wrap around lunch sandwiches. A scary thought considering that in the 1960s, less than 1% of our waste was plastic.

The key problem with plastic however is that a major portion of plastic produced each year is used to make disposable packaging items or other short-lived products that are permanently discarded within a year of manufacture (Hopewell et al. 2009). Well over a billion single-use plastic bags are given out for free every day.

Around 0.2 to 0.3% of plastic production eventually ends up in the ocean (Andrady & Neal 2009). Two of plastics' most touted advantages, their light weight and durability, also make plastic items a significant environmental hazard once seaborne. Close to half of plastics are buoyant and remain so until they become waterlogged or amass too much epibiota to float. Plastics don't biodegrade. Through photodegradation and abrasion plastics only break into smaller and smaller pieces so "that they can be consumed by the smallest marine life at the base of the food web," according to a report by the United Nations Environment Programme (UNEP 2009). Saline marine environments and the cooling effect of the sea mean that degradation requires very long exposure times. Persistence of plastic debris is poignantly illustrated in the account that plastic swallowed by an albatross had originated from a plane shot down 60 years prior some 9,600 km away (Weiss et al. 2006).

Plastics' buoyancy also means they can be easily carried by ocean currents and transported across ocean basins, their contamination stretching from the shorelines to the deepest parts of the sea, from the poles to the Equator and the most remote of islands. Between

1996 and 2006, NOAA recovered 511 tonnes of fishing gear from the reefs of the Northwest Hawaiian Island Marine National Monument (NWHI-MNM), one of the largest marine conservation areas in the world (Pichel et al. 2007). Stewart Island's Mason Bay, located at almost 47° S, is a spectacular, remote and isolated, ca. 10 km sandy beach that is open to the Southern Ocean, facing into the Roaring Forties. The beach is fouled with 2 to 3 tonnes of plastic pollution, mostly fisheries-related items due to intense fishing in close and offshore waters (Barnes et al. 2009). Most of these items are from New Zealand sources. A more minor, but significant, component comes from Korea and Japan; other sources include Argentina, Australia, Belgium, Chile, France, Norway, Poland, Russia, Spain, South Africa, and the United Kingdom (Barnes et al. 2009).

Although most plastic floats on the sea surface, there are an increasing number of reports of sunken plastic debris settling to the sea floor at all depths. A disturbing note is Oshima's (2000, p. 73 in Gregory 2009) report of numerous white plastic shopping bags suspended upside down and freely drifting past a deep-sea submersible at depths of 2,000 m, looking like an assembly of ghosts.

Impacts on ocean wildlife. The bodies of almost all marine species, ranging in size from plankton to marine mammals, and including some of the wildest and most vulnerable species on the planet – animals that make nearly their entire living far from humans – now contain plastic. Sixty percent of 6,136 surface plankton net tows conducted in the western North Atlantic Ocean and Caribbean Sea from 1986 to 2008 contained buoyant plastic pieces, typically millimetres in size (Law et al. 2010). Plastics turn up in bird nests, are worn by hermit crabs instead of shells, and are present in sea turtle, whale and albatross stomachs (Mrosovsky et al. 2009). Over 260 species, including invertebrates, turtles, fish, seabirds and mammals, have been reported to ingest or become entangled in plastic debris, resulting in impaired movement and feeding, reduced reproductive output, lacerations, ulcers, and death (Derraik 2002; Laist 1997).

Entanglement in discarded or lost plastic netting, rope and monofilament lines from commercial fishing is one of the more visible impacts of plastic pollution (Laist 1997). Recent sightings include pods of endangered humpback whales travelling northwards with a mass of tangled rope in tow (e.g., crayfish pot and buoy with marker pole and flag) (Gregory 2009). Lost and abandoned or derelict fishing gear can also continue to capture fish and other species for lengthy periods of time, ("ghost fishing") (Brown & Macfadyen 2007; Goñi 1998).

Ingestion of plastic items occurs much more frequently than entanglement (e.g., Laist 1997; Robards et al. 1997). At sea, plastic bags may often be mistaken for jellyfish, whilst on shorelines

seabirds have been seen to pick at plastic items the same way they pick at cuttlefish bones. In the North Sea, almost all Northern Fulmars (*Fulmarus glacialis*) contain some plastic. Monitoring of plastic loads in seabirds showed increases in plastic ingestion from the 1960s to the 1980s, but have stabilized or decreased more recently (Ryan et al. 2009). On the other hand, microscopic fragments, in some locations outweighing surface zooplankton, revealed a significant increase in abundance when samples from the 1960s and 1970s were compared with the 1980s and 1990s (Barnes et al. 2009). When ingested, such small particles can also be taken up from the gut into other body tissues. Ingestion of plastic can lead to wounds (internal and external); impairment of feeding capacity; blockage of digestive tract followed by satiation and starvation; and general debilitation often leading to death. Plasticizers and organic contaminants that typically sorb and concentrate on plastics at levels far superior to the surrounding marine environment have been shown to affect both development and reproduction in a wide range of marine organisms. Molluscs and crustaceans appear to be particularly sensitive to these compounds (Oehlmann et al. 2009). Being an important food item for many species, plastics ingested by invertebrates then have the potential to transfer toxic substances up the food chain (Teuten et al. 2009). The mechanisms by which ingestion lead to illness and death can often only be surmised because the animals are at sea unobserved or are found ashore dead.

Once fouled with marine life or sediment, plastic items sink to the seafloor contaminating the seabed. Deployment of a remotely operated vehicle submarine in the Fram Strait (Arctic) revealed 0.2 to 0.9 pieces of plastic per km at Hausgarten (2,500 m) (Galgani & Lecornu 2004 in (Barnes et al. 2009)). On dives between 5,500 and 6,770 m, 15 items of debris were observed, of which 13 were plastic (Barnes et al. 2009). The presence of plastic at shallow and greater depths may harm sediment wildlife such as worms, sessile filter feeders, deposit feeders and detritivores, all known to accidentally ingest plastics.

The hard surfaces of pelagic plastics also provide an attractive and alternative substrate to natural floating debris (e.g., seeds, pumice, and wood) for a number of opportunistic colonizers. The increasing availability of these synthetic and non-biodegradable materials in marine debris may increase the dispersal and prospects for invasions by non-indigenous species (Gregory 2009).

Impacts on sea turtles. All sea turtle species are particularly prone and may be seriously harmed by ‘feeding on’ anthropogenic marine debris, particularly plastics (Carr 1987) (e.g., Hawaiian Islands, (Balazs 1985); Texas coast (Shaver 1991); coastal Florida, (Bjorndal et al. 1994); Azores (Barreiros & Barcelos 2001); Western Mediterranean, (Tomás et al. 2002); Paraíba, (Mascarenhas et al. 2004) and Rio Grande do Sul, (Bugoni et al. 2001/, see below) Brazil). Of particular concern are floating plastic bags that might be mistaken for jellyfish, and discarded fishing gear in which sea turtles get entangled, or pieces of which they ingest (Mrosovsky et al. 2009). Laboratory experiments demonstrated that green and loggerhead turtles actively target and consume plastics whether it be small pieces intermixed with food items, or single 1- to 10-cm² sheets (Lutz 1990). Sublethal impacts of plastics on sea turtles can be substantial, yet mortality resulting from interactions with plastic debris is much more difficult to quantify.

Ingestion. Plastic ingestion by sea turtles is a relatively common occurrence, albeit often in small quantities. However, even in small quantities, plastics can kill sea turtles due to obstruction of the oesophagus or perforation of the bowel for example. Relief of gastrointestinal (GI) obstruction of a green turtle off Melbourne beach, Florida, resulted in the animal defecating 74 foreign objects over a period of a month, including four types of latex balloons, five different types of string, nine different types of soft plastic, four different types of hard plastic, a piece of carpet-like material, and two 2 to 4 mm tar balls (Stamper et al. 2009).

Fishing line can be particularly dangerous, when, during normal intestinal function, different parts of the digestive tract pull at different ends of the line. This can result in the gut gathering along the length of the line preventing digesta from passing through the tract (Bjorndal et al. 1994). Plastic ingestion may also indirectly lead to death of an animal through nutrient dilution, i.e., plastic pieces displacing food in the gut (and reducing the surface available for absorption). Typical consequences include decreased growth rates, longer developmental periods at sizes most vulnerable to predation, depleted energy reserves, and lower reproductive output and survivorship of animals (McCauley & Bjorndal 1999). The latter is likely to be an important threat to smaller individuals with a lower ability to increase intake to meet their energetic requirements than larger animals.

Young pelagic sea turtles typically associate with “floating islands” of drifting seaweeds such as *Sargassum*. Floating plastics, tar from terrestrial and oceanic (ship) sources and lost fishing gear are drawn by advection into the same drift lines (Carpenter & Smith 1972; Pichel et al. 2007; Wong et al. 1974). As young sea turtles indiscriminately feed on pelagic material, high occurrences of plastic are common in the digestive tract of these small sea turtles, often contributing to their mortality (Witherington & Witherington 2002).

As plastics can accumulate in multiple segments of the gut, stomach lavages underestimate the incidence of ingestion.

Entanglement. Entanglement in woven plastic sacks, fishing nets, ropes or lines, can prevent sea turtles from diving to feed or from surfacing to breathe. Nets and lines can also amputate limbs, severely reducing an animal’s mobility. Notes on selected studies:

Fifty turtles (23 out of 38 juvenile greens, one out of 10 adult loggerheads and one out of two adult leatherbacks) out of the 92 turtles found dead stranded on the shorelines of Rio Grande do Sul State, Brazil, had ingested considerable amount of anthropogenic debris. Most of this debris consisted of plastic bags and ropes, causing severe lesions and/or obstruction of the digestive tract, linked to the death of four green turtles (Bugoni et al. 2001).

Of 51 sea turtle carcasses that washed ashore in Florida, 25 had ingested debris, which included plastic pieces and fishing lines.

The death of at least two animals was attributed to ingestion of monofilament line (Bjorndal et al. 1994).

Forty one of 54 turtles illegally captured by fishermen in Spain had plastic debris in their digestive tract (Tomás et al. 2002).

Necropsy records of 408 leatherback turtles, spanning 123 years (1885 - 2007), were studied for the presence or absence of plastic in the GI tract. Plastic was reported in 34% of these cases, with a marked increase over time (Mrosovsky et al. 2009).

Hope and the future of plastic in the ocean. “There is a role for individuals, via appropriate use and disposal, particularly recycling; for industry adopting green chemistry, material reduction, and by designing products for reuse and/or end-of-life recyclability; and for governments and policymakers by setting standards and targets, by defining appropriate product labelling to inform and incentivize change, and by funding relevant academic research and technological developments.” (Thompson et al. 2009a).

Re-design. The past decades have proven that there is no stopping the ingenious human mind. Therefore, the development of materials derived from renewable natural resources, with similar functionalities to that of oil-based products, needs to be supported/subsidised. The use of such materials should particularly be encouraged for packaging applications. There is some hope: the Green Chemistry Initiative (Boughton 2009), signed by California Governor Schwarzenegger in 2008, directs the Department of Toxic Substances Control to reduce toxics going into our oceans, including those from plastics, with biodegradable, non-toxic substitutes.

Remove. Beach and ocean cleanups are a great way to raise awareness and to collect data on abundance and trends of debris on shorelines. However, alone they will not solve the problem. At some locations around the world cleaning plastic from the coast amounts to little more than relocation of the items from the beach to inland dumpsites where they pose different problems to the environment and may even find their way back to the ocean when storms or flooding occur. Of concern are high profile “beach cleanups” that serve to mask the severity of the plastic pollution problem with a feel-good event. The most well-run cleanup efforts combine the removal of trash with proper disposal and follow-up educational efforts on how to reduce the production of single-use disposable plastics. When people see and touch plastic pollution they are most open to such behavioural changes.

Reduce, Reuse, Recycle. There is considerable scope for reuse of plastics utilised for the transport of goods, and for potential re-use or re-manufacture of plastic components in goods such as vehicles and electronic equipment (Hopewell et al. 2009). Provided with adequate incentives, industry could be led to use plastic “waste” as raw instead of virgin material, which currently is often cheaper. At much smaller scales users should be encouraged to reuse plastic bags and other plastic goods as much as possible. Although globally only a small proportion of plastics get recycled, mechanical recycling has been increasing at 7% per year in Western Europe (Thompson et al. 2009a). Public support for recycling is high in some countries (57% in the UK and 80% in Australia (Hopewell et al. 2009)). Still, reduction, simplification, and streamlining of everyday packaging, together with clearer labelling could lead to greater separation of materials by users. This would in turn reduce labour associated with sorting costs, currently one of the main impediments to recycling programmes’ efficiency, and maximise the amount that gets recycled, e.g., The Netherlands and Germany.

However, the most efficient and cost-effective solution is to refuse single-use plastic in the first place, and drastically reduce the use of disposable plastic and subsequent release of plastics into the environment. Some simple and immediate actions include:

- o Avoiding plastic-bottled beverages;
- o Buying products with minimal or reusable packaging;
- o Buying in bulk whenever possible to reduce packaging;
- o Buying used items;
- o Seeking out reusable shopping and produce bags like those made from renewable sources (e.g., natural fibres) and always bringing them along;
- o For coffee and or tea – bring your own mug;
- o For food – bring your own container.

Personal actions can advance social change, yet policy actions are oft where the most significant advances are found. For example, Ireland, Eritrea, Rwanda, China, South Africa, Bangladesh, Thailand and Taiwan, have banned or taxed plastic bags. In July 2009, the southern Australian town of Bundanoon became the first community in the world to pass a law banning PET bottles (Malkin 2009). Bans on polystyrene, bottled water and plastic bags are being implemented by communities, businesses and universities around the world, and these trends are expected to continue. At the international level, the United Nations Environment Programme is calling for a worldwide ban on plastic bags.

Continued research on the impacts of plastic on the ocean environment and human health is likely to conclude the problem is worse than currently understood. Plastic production and pollution continues to increase at most locations. The symptom of this growing crisis can be seen inside and on sea turtles as well as their oceanic and terrestrial habitats. Bold initiatives that directly confront the source of plastic pollution, redesign packaging and rethink the very idea of “throwaway culture” are urgently required (e.g., Plastic PollutionCoalition.org). Sea turtle researchers and conservationists have a unique role to play in this cultural evolution, as we have watched the havoc the surge of plastic has caused first hand.

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Editorial: The Zero Counts

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For several years, I have been developing methods to analyze the seasonality of nests or tracks of marine turtles during the nesting season (Girondot & Fretey 1996; Girondot et al. 2007; Girondot et al. 2006; Godgenger et al. 2009). As part of this, I have been in contact with many groups that shared their data for analysis. Often, a similar problem emerged in which the data collectors did not distinguish between a record of zero tracks (or nests) counted and a night (or day) in which no monitoring was conducted. In this paper, I will demonstrate that this lack of precision can create a strong bias as a result of time-series analysis. A methodology is proposed to use nevertheless such a data but with loss of precision.

Calculating the size of a marine turtle population is an essential step toward assessing population status and trends. There are various challenges associated with directly counting the total number of individuals in a marine turtle population, including cryptic life history stages, trans-oceanic dispersal, and non-sequential annual reproduction (Meylan & Meylan 1999). As a result, researchers have traditionally relied on enumerating numbers of nests laid by a population as an index of population size (Gerrodette & Taylor 1999). Because the adult females leave wide deep tracks on the beach after nesting, it is a relatively easy task to identify a sea turtle nesting crawl (Schroeder & Murphy 1999). A challenge in counting sea turtle nests for field workers is that nesting seasons usually span several months and turtles can lay their eggs on remote beaches that are difficult to access. As a result, for many programs engaged in sea turtle nest monitoring, there are often temporal and/or spatial gaps in monitoring effort that must be corrected for, particularly when comparing datasets from different years or populations.

Recently, several methods have been developed to fill spatial or temporal gaps. The general idea is to build a mathematical function that fits the known point and is used to estimate the missing points. The mathematical function can be parametric or obtained nonparametrically by local fitting. During the fit, a measure of distance (least-square or maximum-likelihood) is used to compare each known count with estimated one. During this fit, a zero count value is used exactly in the same way as any other count value and is informative. However, based on my personal experience with various databases from around the world, zero counts are often not reported and thus are potentially confounded with no monitoring for that day (or night). In cases where a zero and a lack of monitoring are not differentiated, the use of statistical methods to fill this gap in nest counts will produce a strong bias.

To demonstrate this effect, I will use a published complete time-series of leatherback nest counts obtained in 2001 in eastern French Guiana (Gratiot et al. 2006). A total of 2762 nests were recorded during daily patrols for the entire year (Figure 1A). For demonstration purposes, I removed all zero counts from this time series and then re-analyzed the revised data with gaps, to get an estimate of the total number of nests during this season (see annex and Girondot 2010). The resulting value was 3295 nests, with a

95% confidence interval between 3262 and 3328 nests (Figure 1B), which is nearly a 20% overestimate. This bias is a direct result of the algorithm, which replaced the missing data (i.e. nights originally with 0 counts) with values derived from the mathematical function.

To better deal with this situation, conditional probabilities must be used. First, recall that likelihood is proportional to the hypothetical probability that an event that has already occurred would yield a specific outcome. Given outcome A , use the likelihood function:

$$L(B|A)$$

to reason about parameters B (B refers to a multidimensional set of parameters and b to one particular set of values for B). The likelihood of N_i nests observed the night i is then:

$$L(B|N_i) = \alpha P(N_i|B=b)$$

with α being a proportionality constant.

From the previous situation, we know that among the events that have already occurred, those with 0 counts are discarded. When the $N_i=0$ are discarded, a new likelihood function will be used:

$$L(B|N_i \neq 0) = \alpha \frac{P(N_i|B=b)}{1 - P(0|B=b)}$$

The set of parameters B maximizing this likelihood is then searched for. With this set of parameters, the final estimate for the total number of nests was 2792 nests, 95% confidence interval between 2762 and 2883 nests (Figure 1C); which is a better approximation of the true total (2762 in Figure 1A).

The new likelihood function allows one to circumvent the lack of information about nights monitored vs. zero nest (or track) counts. However, this estimation is done at the cost of precision of the estimate. The use of this likelihood function implies that all the nights with 0 counts are discarded.

Based on the various statistical tools available, it is not necessary to monitor marine turtle nests or tracks during the entire season in order to generate a good estimate of the nesting effort. However, information concerning the quality of the numbers produced during the monitored night is important. Quality of information is related to:

- Which nights were monitored?
- Does the final count represent all the nests or tracks of the entire night or could have some tracks been missed (e.g. they were erased by high tide)?
- Does the count total reflect the entire beach or only a portion?
- Are there other factors that affect the uncertainty of the final count (species misidentification, confusing true nests with false nests, counting tracks from previous nights)?

Perhaps in the future we may be able to manage all these uncertainties using adequate statistical tools, but for now they should be reported

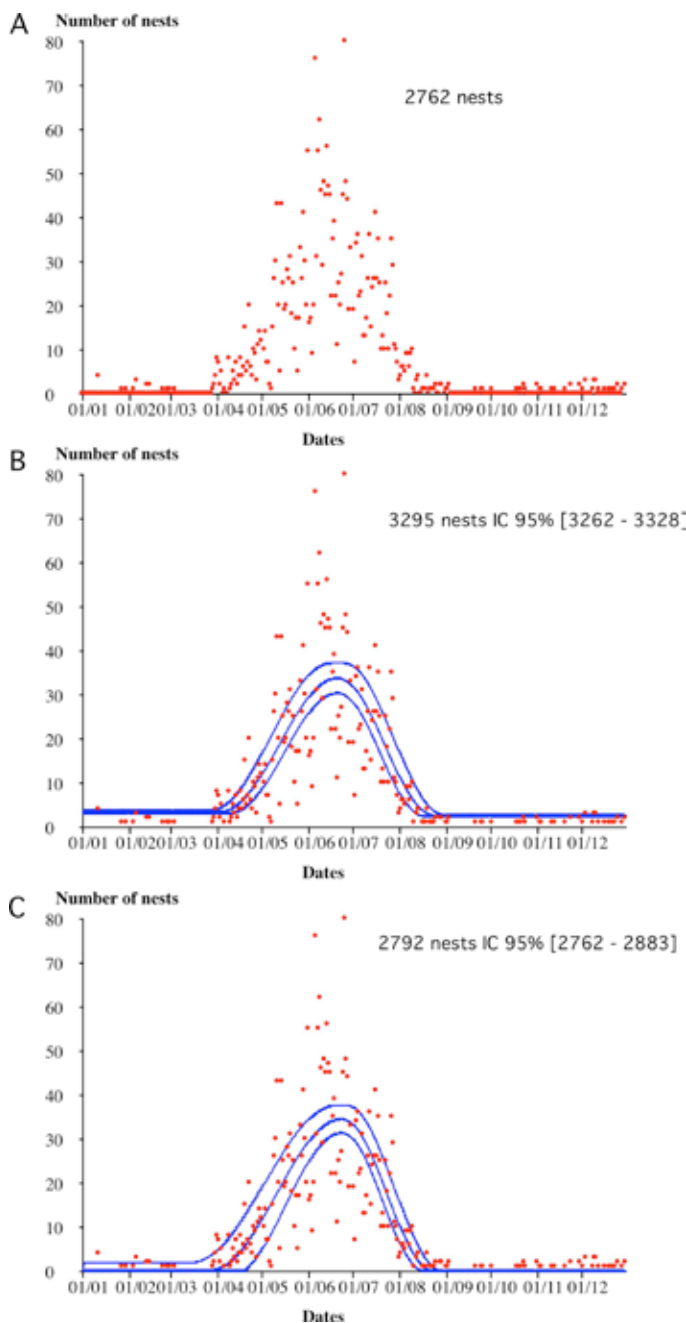


Figure 1. Leatherback nest counts in 2001 in eastern French Guiana, from Gratiot et al. (2006). The blue curves represent the best-fit model with its 95% confidence interval, generated by bootstrapping 1000 times.

by field workers and not hidden.

Finally, there is a similar situation in which nesting effort is reported at the scale of an entire season. A beach that is monitored and zero nests are recorded is not the same as a non-monitored beach. It may seem ridiculous to report a zero nest count for Greenland beaches, as it was to report zero nests along beaches in continental France, before a loggerhead nest was found in 2006 (Sénégal et al. 2009). In the context of global climate change and possible establishment of new sea turtle nesting areas, it may be increasingly challenging to interpret maps where only beaches with >0 nests are reported. Such a situation can be seen in the wonderful maps produced by SWOT (<http://seaturtlestatus.org/learn/maps/all>).

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Annex: Model for marine turtle nesting season analysis (Girondot 2010)

Let t being a day, the number of nests deposited per night is modeled using system of equations:

The model requires at most 7 parameters that have a direct biological interpretation: $MinB$ is the mean nightly nest number before the beginning of the nesting season; $MinE$ is the mean nightly nest number after the end of the nesting season; Max is the mean number of nests at the peak of the nesting season; P is the date of the peak of the nesting season; F is the number of days being flat around the date P ; B is the date of the beginning of the nesting season; E is the date of the end of the nesting season

Likelihood function used here is a negative binomial distribution, which can be described as a combination of various Poisson distributions with different l values (Lawless 1987). The negative binomial distribution has broad applications as a model for count data, particularly those exhibiting overdispersion; i.e. with sample variance exceeding the mean (Lloyd-Smith

2007). In the biological literature, classic uses of the negative binomial distribution include analysis of parasite loads, species occurrence, parasitoid attacks, abundance samples and spatial clustering of populations.

$$\begin{cases} \text{if } t < B \rightarrow \text{Min}B \\ \text{if } t \in [B, P - F/2] \rightarrow \left(\left(1 + \cos\left(\pi(P - F/2 - t) / (P - F/2 - B) \right) \right) / 2 \right) (Max - \text{Min}B) + \text{Min}B \\ \text{if } t \in [P - F/2, P + F/2] \rightarrow \text{Max} \\ \text{if } t \in [P + F/2, E] \rightarrow \left(\left(1 + \cos\left(\pi(t - P + F/2) / (E - P + F/2) \right) \right) / 2 \right) (Max - \text{Min}E) + \text{Min}E \\ \text{if } t > E \rightarrow \text{Min}E \end{cases}$$

Radar Beacon Balls as a Recovery Aid in Field Studies

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Field biologists that follow nest fates face a multifold challenge in repeated visits to the nesting beaches situated in zones of environmental flux or within obvious public domains. A need often arises for cryptic marking methods that do not clue predators or human vandals to an exact nest location (Bowen et al. 1976; Nilson 1985; Vacca and Handel 1988; Tuberville and Burke 1994; Burke et al. 2005). Furthermore, nest locations marked on dynamic beach environments are merely hopeful that storm events do not remodel the beach fore-slopes and dune terraces. Nevertheless, recording the distances and location of nests from multiple physical landmarks remains a standard procedure to document the final outcomes of nest site selection, nest fates, and hatching percentages.

Thus, a general problem statement is a search for technological solutions in fine scale geolocation, especially if involving the recovery of valuable research samples or data loggers. Redundant modes of nest relocation are also desirable for the standard triangulation methods or GPS locations. We report herein on relocation and detection trials for radar beacon balls buried in the sand to test a novel tool for effective recovery of sea turtle nests or data loggers.

Radar beacon balls enable fine-scale detections at 10-33 m distances under open range conditions, but were never intended for underground use or underwater detection. Therefore, field testing was required to specifically evaluate the detection performance at different distances, depths, and substrates (including wet vs. dry seasons and nourished vs. non-nourished substrates) that were challenges represented by sea turtle nesting beaches. Furthermore, the results should be relevant to other field studies seeking a cryptic marking method or tool for fine-scale geolocation.

We evaluated the RadarGolf™ balls and hand-held detectors (Radar Corporation, www.radargolf.com), which are directionally sensitive with a visual bar indicator for signal reception strength, accompanied by an audible pulse that increases as a detector approaches a target ball. The golf balls were regulation size (i.e. 4.2 cm diameter) and not significantly different than the average egg diameter of loggerhead turtle eggs (mean = 4.1 cm, s.d. = 0.1 cm, n = 10, for *Caretta caretta* eggs in Sarasota County, Florida). Balls were not modified except to write an institutional name and contact phone number on the surface. Nests or data loggers were triangulated

in a standard manner with a tape measure and marked each with a Radargolf ball located 1 ft landward from the nest chamber. Golf balls were presumed inert but we evaluated an equal number of nests with and without golf balls to evaluate any possible negative effects of inclusion of a ball on nest hatch rates.

At recovery 2-3 months later, the signal strength indicator and a changing rate of audible pulses pinpointed the vicinity of a buried golf ball. Searches approached from seaward to landward to maximize the sensitivity. Detection units were held perpendicular to the ground and at arm's reach, adjusting the angle slightly as needed when signal sensing to move toward the strongest signal alarm and indication. Distance and signal strengths were recorded as the number of indicator bars at 1 m steps while approaching the nest site.

We evaluated variation in detection distances and burial depths to simulate species ranging from shallow *Lepidochelys* nests to deeper *Dermochelys* nests. In a first experiment, beacon balls were tested on loggerhead nesting beaches of Dauphin Island, Alabama on 9/5/07 at varied depths (5, 10, 15, 20, 25, 30, 35, 40, 50, 60, 70, 80 cm depth) and for three zones used by turtles: upper, middle, and lower zones of the beach width to evaluate nests deposited in those areas or that might later represent a washed out nest scenario.

Distance	Upper Beach				Middle Beach				Lower Beach			
	Depth (cm)				Depth (cm)				Depth (cm)			
	20	40	60	80	20	40	60	80	20	40	60	80
0	9	9	7	4	9	9	5	5	9	9	7	7
1	7	7	3	2	8	4	3	3	8	6	4	2
2	5	4	2	0	5	2	2	2	5	5	2	0
3	4	1	0	0	3	1	0	0	3	3	1	0
4	2	0	0	0	1	0	0	0	2	1	0	0
5	0	7	0	0	0	0	0	0	0	0	0	0

Table 1. Comparisons of depth (20, 40, 60, 80 cm depth) and position along the beach (upper, middle, lower sections of a cross beach profile) on signal detection strength (number of signal bars from 0=low to 9=high) of a buried radar golf ball. Data collected on Dauphin Island, AL, distance in meters.

Distance	Upper Beach Depth (cm)				Middle Beach Depth (cm)				Lower Beach Depth (cm)			
	Nour.		Nat.		Nour.		Nat.		Nour.		Nat.	
	10	20	10	20	10	20	10	20	10	20	10	20
0	9	8	8	4	9	8	7	1	8	1	4	0
1	5	2	5	2	6	2	4	0	5	0	2	0
2	3	0	5	0	4	0	2	0	3	0	0	0
3	1	0	5	0	2	0	0	0	1	0	0	0
4	0	0	3	0	1	0	0	0	0	0	0	0
5	0	0	1	0	0	0	0	0	0	0	0	0

Table 2. Comparisons of nourished (Nour.) and native (Nat.) beach substrates and position along the beach (upper, middle, lower sections of a cross beach profile) on signal detection strength (number of signal bars from 0= low to 9 =high) of a buried radar golf ball. Nourished = Siesta Key, FL. Native = Casey Key, FL. Distance in meters.

In a second test beacon balls were evaluated under selected depths (10, 20 cm) on adjacent beaches that were recently nourished (Siesta Key, Florida) and native non-nourished beaches (Casey Key, Florida) to grossly evaluate differences in substrate types and water content across the upper, middle, and lower beach zones (8/13/07). The signal reception tests at selected depths (10, 20 cm) were repeated on Casey Key to contrast a dry substrate at the end of a dry season (5/10/07) with a more saturated substrate after the hurricane season was well underway (8/22/07).

As a general test of recovery efficiency, a regional scale evaluation was conducted for the recoveries of golf balls buried adjacent to multiple data loggers deployed separately on 11 beaches (Bon Secour, Panama City, Cape San Blas, St. George Island, Lido Beach, Casey Key, Venice, Boca Raton, Juno Beach, St. Lucie, Melbourne). Data loggers were deployed early in the nesting season (June) as part of a thermal study and a golf ball was buried adjacent to the data logger. When data loggers were recovered at the end of the season (November), the time taken with golf balls as a relocation aid was compared with a timed triangulation measurement from marker stakes. As a double blind, the person relocating the golf balls was naïve (i.e., not the same as the June person) to exclude any use of visual cues and evaluate the device solely on the alerts of the detector. The recovery times were compared with a golf ball versus the time to find data loggers by tape measure (method was chosen at recovery by a coin flip).

Nests with golf balls (both inundated and non-inundated) had no substantive difference of hatching success at 68.5%, compared to a combined hatch success of 61.1% from *in situ* hatch success, derived 38.4% with inundated nests and 83.7% with non-inundated loggerhead nests monitored on five islands (Longboat Key, Lido Key, Siesta Key, Casey Key, Venice Island) in the same season.

Representative data for upper, middle, and lower beach zones of Dauphin Island, AL (Table 1) indicated no substantial difference among beach zones, but an obvious decline in detection distance with depth of burial. All balls could be detected within 2 m regardless of depth of burial. The deepest buried balls had weaker signal strengths and required a closer approach before detection. However, all balls gave reliable signals that were easily approached and allowed the tester to be within a 1m patch, usually pinpointing a buried ball.

Distance	Upper Beach Depth (cm)				Middle Beach Depth (cm)				Lower Beach Depth (cm)			
	Wet		Dry		Wet		Dry		Wet		Dry	
	10	20	10	20	10	20	10	20	10	20	10	20
0	8	4	6	3	8	4	6	1	4	0	6	1
1	5	2	4	1	5	2	5	2	2	0	4	1
2	5	0	3	0	5	0	3	0	0	0	2	0
3	3	0	1	0	3	0	1	0	0	0	0	0
4	1	0	0	0	1	0	0	0	0	0	0	0
5	0	0	0	0	0	0	0	0	0	0	0	0

Table 3. Comparisons of substrate wetness and position along the beach (upper, middle, lower sections of a cross beach profile) on signal detection strength (number of signal bars from 0= low to 9 =high) of a buried radar golf ball. All data collected on Casey Key, FL. Distance in meters.

Detection distances were greater on a recently nourished beach than a non-nourished beach (Table 2). A comparison of dry and wet seasons indicated that detection distances were greater in wet sand than dry sand for the upper and middle sections of beach but reversed to be greater detection distance in dry than wet sand for the lower beach section (Table 3).

With recovery method as a fixed factor and depth and distance as covariates in an ANCOVA, there was no significant difference in mean recovery time for data loggers found by golf balls (143 sec) or tape measure (191 sec) ($F_{1,49} = 0.11, P = 0.75$), although differences were noted by ball depth ($F_{1,49} = 8.76, P = 0.005$) but not by distance from the vegetation line ($F_{1,49} = 2.67, P = 0.11$). Confounding factors appeared to be that either method was suitable in stable upper beach zones, but for mid and lower beach positions there was less of a signal so the ball was slightly harder to find. This finding was dissected by the following tests. For 87 data loggers marked by golf balls on urban beaches across multiple Florida beaches, 87% of the golf balls were recovered, 8% were found by other persons on the beach and reported, and 5% were not found at all. For 50 nests marked for recovery in Sarasota County, 70% of golf balls were recovered, and 30% were not found at all, whether from nests washing away or human tampering.

Briefly, the test outcomes suggested that: (a) even balls at 80 cm depth could readily be found within a 30 cm radius, (b) lesser depths still yielded wider detection ranges (up to 5 m), and (c) detection was minimally affected by different beach substrates or substrate moisture. (d) Detection distances were similar in upper and middle beach zones and decline slightly in lower beach positions but not substantially so. (e) Radio beacon balls had no detectable effect on nest viability or hatch success. (f) There was no substantial or at best a moderate time savings over regular triangulations in relocation of data-loggers or nests. However it is noteworthy that for situations when the triangulation markers (i.e. stakes) were no longer present when we returned to retrieve data loggers, the buried beacon balls were present and often facilitated their recovery anyway.

We observed that urban beaches may have increase possibilities of human tampering if golf balls are buried shallowly. Nests that are entirely washed away, as opposed to inundated nests, will probably lose the golf ball marker. Thus, the devices are a convenience, but not a panacea and still vulnerable to loss from erosion or humans.

The range of depths, likelihood of marker stake loss and nest erosion, and potential for human tampering at a given beach will determine whether radar beacon balls prove to be useful recovery aids in other circumstances for other beaches.

Retail costs of an initial detection device and a dozen Radargolf balls were \$200US and additional balls were \$40US per dozen. However, package discounts can occasionally be found, and demo balls can be cheaper if bulk ordered directly from the manufacturer. A next generation device was under development that may be worth evaluating in the future.

In comparison to alternative methods, triangulation and stakes or other permanent markers are simple and cheap and should always be a default method. Hand held GPS units can range from \$100-300 depending on the model, but are still limited in accuracy. Surveyor quality GPS is more expensive and requires a substantial learning curve. Other expensive methods include ground penetrating radar at costs upwards of \$30,000 without software, but can be rented for \$250 a day (Stott 1996, Kinlaw et al. 2007). Recent developments with PIT tag and PIT-pack detection units still are showing a severe limitation on detection distance (Blomquist et al. 2008).

In contrast, the radio beacon balls were relatively cheap, quick, and with a trivial learning curve. The detection units were rugged and reusable. There is no battery within a ball so each ball is reusable across multiple seasons. Carrying a few golf balls in a pocket or daypack is convenient as a backup method to mark nests or data loggers that will be returned to later. In our experience, inexperienced users can intuitively understand the concept and operate in less time than it takes to explain, typically less than 30 seconds. Researchers needing a rapid means of relocating buried data loggers or nests may find the device a useful addition to their tool-bag of field marking techniques. For situations where a more cryptic marking method is desirable that does not give clues to predators or invite random vandalism, beacon balls may be useful tool. However, a convenient backup aid cannot function reliably as the sole recovery method for nest locations or data loggers if human tampering or severe erosion are factors of local concern.

Researchers operating in terrestrial conditions will undoubtedly find more favorable results than the severe tests represented here for dynamic beach environments.

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Olive Ridley Turtle Presence and Nesting Records for Western Australia

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Prince (1993) noted that little was known about the distribution of olive ridley turtles (*Lepidochelys olivacea*) in Western Australia (WA), although they appeared to frequent the far northern coastal waters in small numbers. There were no nesting records of this species in WA, in part due to the remoteness of areas that contained likely suitable habitat. This note reports on subsequent observations of olive ridley turtles at sea off the WA coastline and of the first scientifically documented nesting of olive ridleys in WA.

The presence of feeding turtles at sea has been confirmed from four northern WA at sea encounters with apparent sub-adult or adult

size turtles as a result of networking with industry and community. These records comprise two capture and live release cases from fisheries, one from an inshore prawn trawl, and the other from a longliner working off the Pilbara coast.

A third turtle was recovered by recreational fishers at sea north of Broome while suffering from 'floating syndrome' and presented to a registered wildlife carer for attempted rehabilitation. The fourth turtle had been fatally entangled in floating marine debris and was reported by offshore oil industry personnel on the FPSO Northern Endeavour in the Timor Sea. All four cases have photographic

ID	Location	F-primer	R-primer	Haplotype
WAM R169921 = DEC BP443	Darcy Islands, nest #2	LTEi9	H950	LOIP-02
WAM R169918 = DEC BP439	Darcy Islands, nest #1	LTEi9	H950	LOIP-01
WAM R169923 = DEC BP2068	Langgi Beach, near Stone Warriors	LTEi9	H950	LOIP-01
WAM R151598 = Hatchling 2008	Cape Leveque, near Kooljamon Resort	LTEi9	H950	LOIP-03

Table 1. Samples results from the four Western Australian olive ridley hatchlings sequenced at their mtDNA control region haplotypes

records and case notes that can be viewed at:

<http://www.seaturtle.org/cgi-bin/imagelib/index.pl?cat=500&user=3607&thumb=1>.

Additionally, a stranded carcass was collected that is believed most likely the result of a fatal trawl capture somewhere off far northern Australia, with the carcass being retained frozen on board. Later translocation to and subsequent dumping to sea in the Perth region is suspected from a vessel home-porting in this area but usually fishing far away. A separate case report will be compiled for this turtle. Skeletal remains are registered with the WA Museum - R125596. Measurements of the carcass were: curved carapace length (CCL) 665 mm; curved carapace width = 650 mm; mass = 26.2 kg. Tail length (not recorded) suggested the turtle was male.

In addition to these at sea records, four nesting events by olive ridley turtles were documented from the Kimberley region in 2008 (3), and 2009 (1); the first of 2008 on the west Kimberley coast near Cape Leveque; and the others from Darcy Island in the Bonaparte Archipelago, northwest Kimberley, WA. The 2009 record was also from northwest Kimberley, WA, on the Langgi coast at the tourist location known as The Stone Warriors. These cases have photographic records, are specimen backed, and species identity has also been confirmed by species-specific genetic tests.

The Cape Leveque nesting record is from a single nest emergence discovered 23 April 2008 by visitors to the Kooljamon Resort. This comprised a group photograph of hatchlings at the nest site, photographs of one intact specimen being retained by the Bardi Jawi Rangers group, and remains of one mummified hatchling and 6 unhatched eggs excavated several weeks later from the same nest site as recorded by GPS. Photo scaling suggests that the straight carapace length (SCL) of the retained specimen was ~36 mm; the mummified hatchling SCL was ~37 mm. Maximum diameter of the unhatched eggs varied from 36.6 - 39.4 mm (mean 38.2 ± 0.4 mm SE). All eggs were later sacrificed in unsuccessful attempts to extract DNA samples.

The Darcy Island observations were made on 4 and 6 July 2008. Hatchlings from three nests were recorded. These comprised: a photograph only of one live hatchling from nest #3 excavation on 6 July; one retained specimen from nest #1 and four from nest #2 during emerged nest site excavations on 4 July. Field measurements were not taken. SCLs of the preserved hatchlings varied from 39.0 to 42.5 mm (grouped mean 40.3 ± 0.7 mm SE).

The Langgi specimen was recovered late May 2009 at a nest site apparently disturbed by a dingo, but from which its siblings had apparently already escaped.

Discovery times for the Kimberley hatchling samples recorded here are consistent with the pattern and timing of olive ridley nesting reported for the Tiwi Islands, NT, by Whiting *et al.* (2007).

Ventral scapulation patterns of all hatchlings that could be

determined included four inframarginal or bridge scales. The expected inframarginal pores could not be visualised.

Dorsal scapulation was variable. The 1st costals contacted the nuchal in all cases. The maximum regular bilateral costal pattern was seven pairs. Other patterns included sub-divided or inserted scutes within basic six or seven pair patterns. Prefrontals were paired. These dorsal scale patterns are evident on the images that are posted at:

<http://www.seaturtle.org/cgi-bin/imagelib/index.pl?cat=500&user=3607&thumb=1>.

Genetic analyses. DNA was extracted from two Darcy Island hatchling specimens originating from two different nests (WAM R169921 = DEC BP443 and WAM R169918 = DEC BP439), from the mummified hatchling (WAM R151598 = Hatchling 2008) from the Cape Leveque nest, and from the hatchling from the fourth nest at Langgi (WAM R169923 = DEC BP2067). A 780 bp fragment of the mtDNA control region was amplified using primers LTEi9 and H950 (Abreu-Grobois *et al.* 2006). PCR product was checked for quality and quantity and purified prior to sequencing and negative controls were used to check for contamination. All samples were sequenced in both directions to confirm the results. The sequences were then compared with known olive ridley sequences from Australian nesting populations on Tiwi Islands, Northern Territory, the McCluer Group, Northern Territory, and Flinders Beach, western Cape York Peninsula, Queensland (Jensen *et al.* unpublished data).

Three haplotypes were found among the four samples and all of them were identified as being olive ridley sequences. The two Darcy Island samples had haplotypes LOIP-01 and LOIP-02. The one sample from Langgi also had haplotype LOIP-01, which is the most common haplotype found in ~75% of turtles nesting in the Northern



Figure 1. Northern Australian nesting locations for olive ridley turtles with associated haplotype data.

Territory at the Tiwi Islands and the McCluer Islands group (Jensen *et al.* unpublished data). Haplotype LOIP-01 corresponds also to the short 470 bp haplotype J reported by Bowen *et al.* (1998). Haplotype LOIP-02 has been found at all three nesting populations sampled across northern Australia, but at low to intermediate frequencies (5 – 30 %). Interestingly, the sample from Cape Leveque had the haplotype LOIP-03. This haplotype is found in high frequency on western Cape York (Flinders Beach), Queensland, but only in very low frequency in the Tiwi Islands and is absent from the McCluer Group nesting population (Jensen *et al.* unpublished data). All three haplotypes above have been reported from Australian nesting populations only (Table 1). Locations mentioned in the text are shown in Figure 1.

Information provided in this paper supports the view that the olive ridley turtle is part of the northwestern Western Australian region marine turtle fauna, having both nesting and foraging presence.

Acknowledgments: We thank Ben and Alexis Dodwell for reporting the Kooljamon nest emergence to the Bardi Jawi Rangers; Roy Teale for the Darcy Islands collections; and Emma Francis for the Langgi specimen. Nancy FitzSimmons' comments on the text in preparation were most helpful. Photographic records of the at sea discoveries posted on the seaturtle.org Image Library are annotated as acceptable to the providers, for whom we are most grateful. The Bardi Jawi Rangers are part of NAILSMA and

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Long Distance Migrations of Hawksbills Tagged as Juveniles at Aldabra Atoll: Confirmation from Digital Photography

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Since the 1950s, the flipper tag has been a standard research tool of sea turtle biologists, its key features being a 'unique' identification code on the front of each tag and a return address on the back (http://www.ioseaturtles.org/flipper_background.php). Much of what we know about long distance turtle migrations derives from data obtained when turtles, tagged on a nesting beach or foraging ground, are subsequently observed (either dead or alive) at distant feeding or nesting sites, and the tags are recorded and/or removed and returned to the address on the back of the tag.

A possible confounding factor when reporting tag returns is transcription error. For instance, when tagged turtles are encountered underwater by divers, removal of the tag is usually undesirable or impossible. The observer records the identification code, and the tagged turtle continues on its way. But, it may be difficult for a diver to accurately read the tag. The identification codes of turtle tags typically comprise six to eight characters, and if any one of them is misread or misrecorded the presumed identity of the turtle will be false and the data misleading. Unfortunately, such errors are a major problem for many tagging programs (JAM, personal observation).

Digital photography can provide the verification needed to confirm the identity of an unusual tag. Digital cameras or mobile phones

capable of taking photos on land are widely used and relatively inexpensive, and many divers are now armed with underwater digital cameras. By routinely photographing unusual tags on nesting turtles and any tags encountered on turtles underwater, unexpected tag recoveries can be confirmed with certainty. Additionally, digital photos can sometimes reveal and record other information of interest.

We report on two long distance tag recoveries involving hawksbills (Turtle 1 and Turtle 2) that were originally tagged as juveniles at Aldabra Atoll, Seychelles in 1997 and 1996, and resighted 9.6 and 13.2 years later at distant localities 900 and 1,150 km away, respectively (Table 1, Fig. 1).

Turtle 1. On 18 November 2006, off Anse Mondon, Silhouette Island, Seychelles, an observant SCUBA diver with a camera took photographs of a tagged hawksbill showing: a) view of the entire turtle taken perpendicularly to its central axis from the tagged right side (Fig. 2); and b) close-up of the titanium tag itself on the right front flipper (---1756) (Fig. 3). The prefix letters were not clearly visible, but 'SEY' was the only prefix used on Titanium tags in Seychelles at the time the photo was taken. Nevertheless, other possible origins of the tag were investigated and ruled out

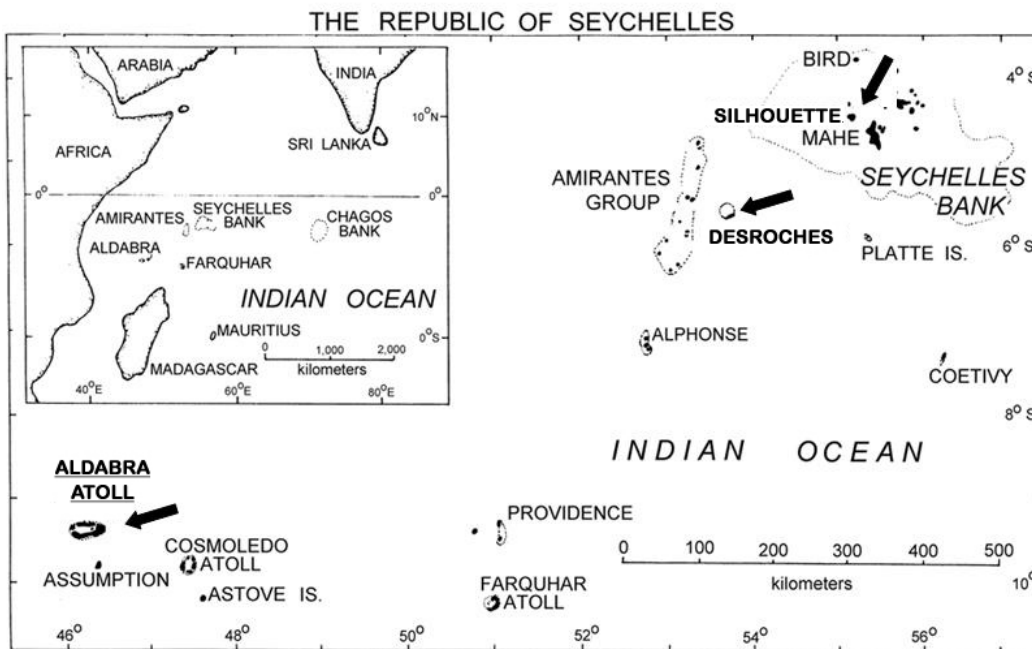


Figure 1. Map of Seychelles indicating locations of Aldabra Atoll (tagging site), and Desroches and Silhouette Islands (localities of subsequent sightings)

by examining the flipper tag inventories of the Archie Carr Center for Sea Turtle Research (<http://accstr.ufl.edu/taginv.html>) and the Indian Ocean South East Asian Marine Turtle MoU (<http://www.ioseaturtles.org/flippertags.php>), by posting the discovery of the tag on the CTURTLE listserv, and through correspondence with project leaders working in the Indian Ocean region. These efforts concluded that the turtle had been tagged as 'SEY1756' in 1997 (9.6 years earlier) in shallow water near Ile Esprit, a small island inside the inner lagoon of Aldabra atoll, 1,150 km to the south west. When tagged, the turtle had a straight carapace length notch to tip (SCLn-t, as defined by Bolten 1999) of 56.4 cm and weighed 21.5 kg. In contrast, the turtle in the 2006 photo (Fig. 2) is clearly a



Figure 2. Digitally photographed full lateral view of Turtle 1 indicating that it is a subadult male. The most proximal large flipper scale on the right flipper is clearly visible. (Photo credit: Hakkim Jaballah.)

sub-adult male with a developing tail. The Adobe Photoshop CS3 'Measure Tool' was used to calculate from the photos, ratios of: a) tag length to length of the most proximal large flipper scale (in Fig. 3); and b) length of the most proximal large flipper scale to SCLn-t (in Fig. 2). With these figures and knowledge that a Stockbrands Titanium Turtle Tag is 4 cm in length, the SCLn-t of the turtle was calculated to be approximately 70 cm. It follows that between 1997 and 2006 the turtle would have grown approximately 13.6 cm in carapace length - an average of ~1.4 cm/yr.

Turtle 2. On 2 December 2009, an adult female hawksbill bearing tag numbers SEY1151 and SEY1152 was found nesting at Desroches Island in the Amirantes Group of Seychelles. Digital photos taken of

the tags (Fig. 4) confirmed that the turtle was the same one tagged 13.2 years earlier in the shallow waters of Passe Hoareau just inside the inner lagoon of Aldabra Atoll, 900 km south west of Desroches Island. When tagged in 1996, the turtle had a CCLn-t of 57.2 cm and weighed 16.5 kg. In 2009, her CCLn-t measured 86.2 cm; and she dug two pits and laid 111 eggs. Thus, between 1996 and 2009 she grew 29.0 cm in carapace length -- a minimum average growth rate of ~2.2 cm/yr (minimum because we don't know in which year she attained adult size).

These are the only known sightings of Turtles 1 and 2, and the first recorded migrations away from the atoll for any of approximately 500 juvenile hawksbills tagged at Aldabra since 1986 (Seychelles



Figure 3. Close up of the Stockbrands Titanium Turtle tag (---1756) immediately adjacent to the most proximal large scale on the right flipper. The prefix letters are not clearly visible, but were determined to be 'SEY' (from Seychelles) based on communications that ruled out an origin elsewhere in the region. (Photo credit: Hakkim Jaballah.)



Figure 4. Digital photos of tags SEY1151 and SEY1152 on Turtle 2. (Photo credit: P. Summerton.)

	Turtle 1		Turtle 2	
	Initial Tagging	Recent Sighting	Initial Tagging	Recent Sighting
Right flipper tag	SEY1756	SEY1756	SEY1151	SEY1151
Left flipper tag	SEY1757	unknown	SEY1152	SEY1152
Date	20-Apr-97	18-Nov-06	15-Oct-96	2-Dec-09
Locality	Aldabra Atoll, near Ile Esprit	Silhouette Island, off Anse Mondon	Aldabra Atoll, Passe Hoaearu	Amirantes Group, Desroches Island
Habitat	Shallow (1-3 m) sand lagoon	Reef slope	Shallow (<2 m) sea grass, sponge, hard coral reef flat	Nesting beach
Activity	In-water	In-water	In-water	Nesting
Sex	Unknown	Subadult male	Unknown	Adult female
Carapace length	56.4 cm	~70.0 cm	57.2 cm	86.2 cm
Mass	21.5 kg	na	16.5 kg	na
Observer	J.A. Mortimer & T. Jupiter	J. Gerlach & H. Jaballa	J.A. Mortimer & F. Sophola	P. Summerton
Km btwn sightings	1150 km		900 km	
Yrs btwn sightings	9.6 yr		13.2 yr	
Carapace growth	~13.6 cm		29 cm	
Growth rate	~1.4 cm/yr (mean)		~2.2 cm/yr (min)	

Islands Foundation, unpublished data). At the atoll itself, about one third of tagged juvenile hawksbills are subsequently recaptured (Mortimer et al., 2003; Seychelles Islands Foundation, unpublished data). Without the confirmation provided by the digital photos, however, doubt would have persisted as to the true identities of the two animals recorded in such unusual circumstances. Our data demonstrate: a) the value of photographing any unusual turtle tags; b) the extent to which hawksbills of various life stages are a resource shared amongst the far-flung islands of Seychelles (and beyond); and c) the need for national and regional cooperation in the protection of the species.

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Table 1. Detailed information relevant to initial tagging and recent sighting of Turtle 1 and Turtle 2

Identifying and Mitigating Hatchling Disorientation on Nesting Beaches

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Sea turtle hatchlings use mainly visual cues to find the ocean after emerging from the nest during the night (Lohmann et al. 1997). On nesting beaches, artificial lights can disrupt hatchling seafinding and thus are potentially a major threat. If disrupted between the nest and the ocean, hatchlings may be more susceptible to mortality associated with exhaustion, dehydration, predation, etc. (Whiterington & Martin 2000). Similarly, artificial lights can disorient adult females while they are crawling up the beach to nest (or during the nesting process) (Whiterington & Martin 2000; Deem et al. 2007). Identification and quantification of light impacts on the beaches is an important conservation measure in nesting areas (Whiterington & Martin 2000).

The northern coast of Bahia is a major sea turtle nesting area in Brazil, hosting approximately 6,000 nests per year laid by loggerhead (*Caretta caretta*), hawksbill (*Eretmochelys imbricata*), olive ridley (*Lepidochelys olivacea*) and green turtles (*Chelonia mydas*). Conservation activities on nesting beaches in Brazil began in 1982 and continues today, being carried out by Projeto TAMAR (Brazilian National Sea Turtle Conservation Program) (Marcovaldi & Marcovaldi 1999).

Recently, increasing nest numbers have been observed for loggerheads (Marcovaldi & Chaloupka 2007), hawksbills (Marcovaldi et al. 2007) and olive ridleys (Silva et al. 2007) in northern Bahia. Concurrently, tourist activities have also increased in the region, resulting in the development of villages and small cities along the coast, in addition to the construction of large resorts in front of nesting beaches (Lyrio 2003). This coastal development has greatly contributed to the increasing occurrence of artificial light on the beaches. In this study, we describe a simple and efficient method to identify emerged nests with hatchlings disrupted by artificial lighting and locate the source of the disruption, with experimental data obtained from several beaches of northern Bahia.

During the nesting seasons of 2006/2007, 2007/2008 and 2008/2009, we investigated hatchling behavior on four beaches in northern Bahia, Northeast Brazil. Two (Busca Vida or BV and Santa Maria or SM) are located in residential condominiums areas, with households on average about 40 m from nests and occupied during the summer, which is the period for the nesting season. BV also has a tourist resort. The other two beaches (Arembepe or AR and Berta or BE) are located in isolated areas, with no inhabitants or direct light sources reaching the beaches. All beaches were daily patrolled at early morning during the nesting and hatchling emergence season (September to April), according to standard methodology for fieldwork, described in Marcovaldi & Marcovaldi (1999), where all nests were marked (Fig. 1A).

To assess and document the impacts of artificial lights in each of the beaches, we examined hatchling tracks from nests in the early

morning after primary nighttime emergence occurred. If the majority of observed hatchling tracks went to the ocean, we score the nest as “right”. If the majority of observed hatchlings tracks did not go in the direction of the ocean, the nest was scored as “wrong” (Fig. 1A). When live hatchlings were found near the houses in the early morning, we promptly released them. For the three seasons of our study, we observed that on the uninhabited beaches (AR and BE), as expected, 100% of observed nests were scored as “right” (Table 1). However, on the developed beaches (BV and SM), some nests were scored as being “wrong” (Table 1). Overall, SM had a higher rate of “wrong” nests than BV, likely due to the higher density and closer proximity of houses to the nesting beach in SM.

To identify the possible artificial lighting sources causing the observed hatchling disruption, we collected data on the average direction(s) that the hatchlings crawled away from the ocean of each nest (Figure 1B). Using this information, on the following evening, biologists visited the beach with the aim of identifying which light sources attracted the hatchlings. Once identified, we approached these homes or resorts to speak with the owners or managers, to inform of the impacts of their lights on the turtles and provide them with possible actions to eliminate these impacts. Brazil, there is specific legislation prohibiting the impact of artificial lighting on sea turtle nesting beaches (legislation IBAMA Portaria n° 11 of 30th January 1995 and Bahia’s State Law n° 7.034 of 13th February 1997). These laws prohibit the incidence of light on nesting beaches (IBAMA’s federal law applies to specific regions of the Brazilian

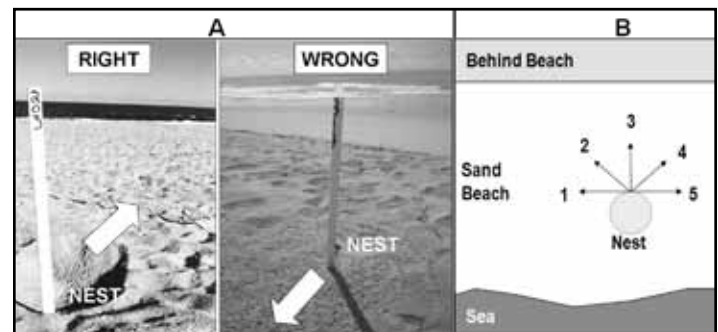


Figure 1. (A) Pictures showing the hatchling’s crawls on the sand at the northeast coast of Bahia. RIGHT is the situation when the majority of hatchlings went toward the sea and WRONG when they went on the opposite direction. (B) Schematic representation of the methodology used for the identification of the track directions (1 to 5) of the hatchling’s crawls disoriented due to a light pollution sources for each disrupted nest.

		BE	AR	SM	BV
Total nests		215	158	407	720
2006/	Nests with hatchling disorientation	0	0	20	6
2007	% nests with hatchling disorientation	0	0	5	0.8
	% light pollution sources identified	n/a	n/a	100	100
Total nests		208	120	406	752
2007/	Nests with hatchling disorientation	0	0	26	18
2008	% nests with hatchling disorientation	0	0	6	2
	% light pollution sources identified	n/a	n/a	100	100
Total nests		344	171	530	803
2008/	Nests with hatchling disorientation	0	0	36	18
2009	% nests with hatchling disorientation	0	0	7	2
	% light pollution sources identified	n/a	n/a	100	100

Table 1. Total number of sea turtle nests (including loggerhead, hawksbill, olive ridley and green) laid on Berta (BE), Arembepe (AR), Santa Maria (SM) and Busca Vida (BV), during the 2006/2007, 2007/2008 and 2008/2009 nesting seasons, together with nests observed with disrupted hatchling sea finding.

coast and Bahia state law applies to specific beaches in the state) from any artificial light source in a range of 50 m from the high tide line.

We also georeferenced each nest with disrupted hatchlings, to enable the visualization of satellite images (i.e. available in Google Earth® software). This helped to identify not only the sources of light that caused hatchling disruption of individual nests but also potential hotspots of artificial light affecting many nests (Fig. 2); it also produced maps there were valuable in raising awareness of lighting issues in public meetings with the property owners and managers.

Overall, the method of scoring nests as “right” or “wrong” was simple and quick, and thus easily integrated into the routine morning patrols of beach monitors. Once identified, nests with disrupted hatchling behavior could then be investigated in more detail, and possible sources of artificial light identified.

Despite the low frequency of nests with disrupted hatchlings at SM and BV (<8%), it should be inferred that there is little hatchling disruption by artificial lights along the entire Bahia coast. This is because there is a wide variety of patterns of development behind beaches across the state; we recommend that each beach should be evaluated independently for hatchling disruption.

Our initial success with this simple method of identifying artificial lighting problems on our beaches is promising. However, it may have implications under certain conditions. For instance, moon phase can play a role in hatchling disruption from artificial lighting (Salmon & Witherington 1995). Thus, it may be necessary to continually check for disruption of hatchlings from nests across seasons and perhaps across years, to fully identify problem lighting sources. However, our method is simple and quick enough to implement as a routine measurement during morning patrols.

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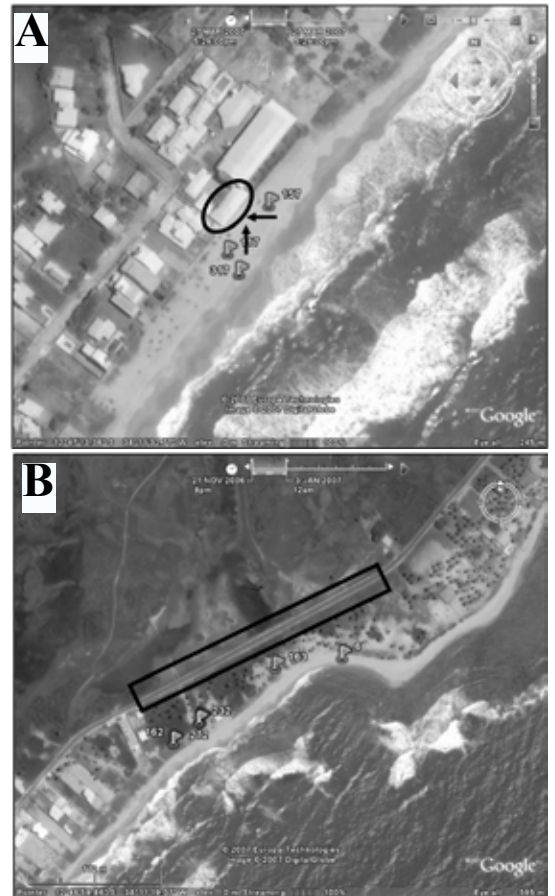


Figure 2. Satellite images used for educational purposes showing examples of light pollution affecting sea turtle hatchlings at Santa Maria beach, northeast Brazil. (A) Situation where some nests, each identified as a numbered flag, had hatchlings that crawled toward (arrows) the same small hotel (circled building). (B) A stretch of beach (surrounded by the box) where the light from public street lamps focused direct on nests (flags numbered) on the beach and hatchlings from several nests had crawled toward the street.

Matthew Godfrey and an anonymous referee for their valuable contributions to the manuscript. Projeto TAMAR, a conservation programme of the Brazilian Ministry of the Environment, is affiliated with the ICMBio (the Brazilian Institute for Biodiversity Conservation – Instituto Chico Mendes), is co-managed by Fundação Pró-Tamar and is officially sponsored by Petrobras.

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Indian Ocean Crossing by a Juvenile Hawksbill Turtle

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The following tag report describes the longest recorded migration for a hawksbill turtle (*Eretmochelys imbricata*) and the first trans-Indian Ocean crossing reported for any sea turtle species.

A juvenile hawksbill turtle with tags CA7443/CA7444 was initially captured as part of a larger mark-recapture study in a seagrass and algal feeding area near South Island, Cocos (Keeling) Islands, Indian Ocean (12° 11.528'E / 96° 54.910'S) on 10 Jan 2003. The Cocos (Keeling) Islands is an external territory of Australia. The turtle appeared healthy and measured 54.7 cm curved carapace length, 44.9 cm curved carapace width and weighed 13 kg. On 22 Sep 2008, it was found dead in a fishing net in the Lindi district of Tanzania on the east coast of Africa (approx. 9° 50' S / 39° 54' E)

over 6100 km straight-line distance from its initial capture position (Fig 1.). No other information was available despite contact with the reporter of the information in July 2010 when the tag recovery was first reported. There was no information to indicate if this turtle was a juvenile or adult when found and therefore deduce if this was a breeding or developmental migration. However, based on growth studies of hawksbill turtles from Cocos (Keeling) Islands (Whiting unpublished), it is unlikely this turtle would not have reached mature size during the 5.7 years at large. If this was a movement made by a juvenile turtle it supports previous evidence of long distance and trans-ocean crossings by hawksbill turtles from tag recoveries (Marcovaldi & Filippini 1991; Bellini *et al.* 2000; Grossman *et al.* 2007) and genetic studies (Bowen *et al.* 2007) in the

Atlantic. In addition, in 2003 a resident juvenile turtle from the Cocos (Keeling) Islands tracked by satellite telemetry made a westward journey of over 1000 km into the middle of the Indian Ocean before transmissions stopped (Whiting & Koch 2006). This current tag recovery and the previous tracked movement indicate that oceanic movements by foraging hawksbills from the remote Cocos (Keeling) atoll may not be unusual. These records from both the Atlantic and the Indian Oceans indicate that juvenile hawksbill turtles may not always remain in the neritic zone once they appear in shallow foraging areas and have the ability to change foraging locations over very large distances. This tag recovery is a substantial movement record for hawksbill turtles and for the Indian Ocean.

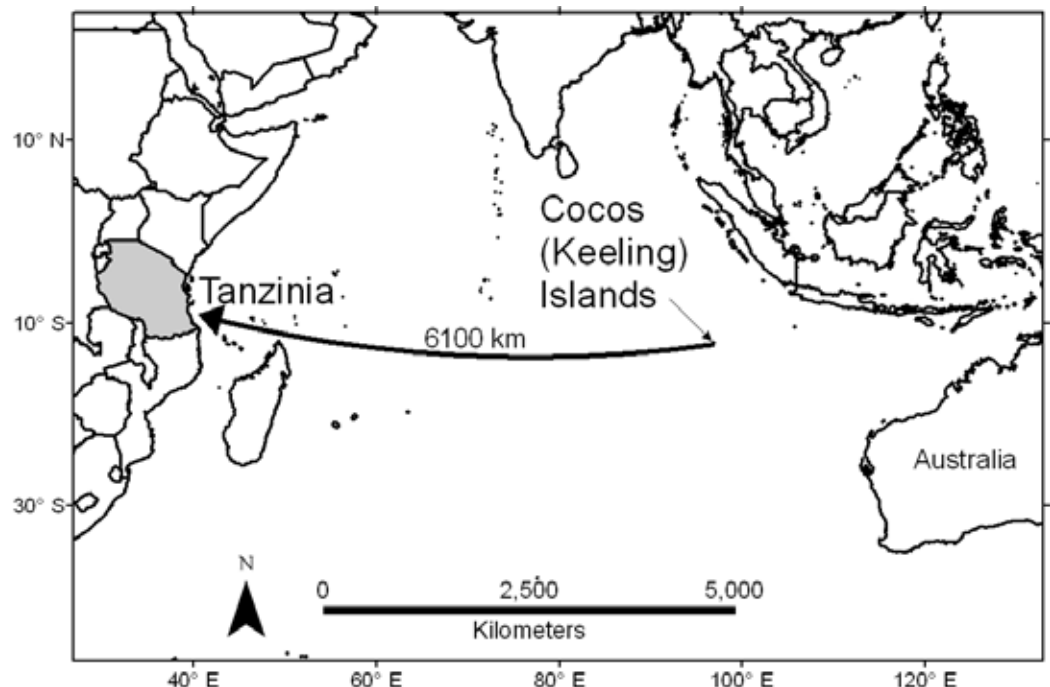


Figure 1. Map illustrating the two capture locations and the minimum distance travelled.

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Using Google Earth to Confirm Marine Turtle Nesting: The Makran Coast (Pakistan) & The Hadramout (Yemen)

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Some 20 years ago, two colleagues and I published results of brief ground and aerial surveys designed to investigate the extent of marine turtle nesting on the coast of Balochistan Province, Pakistan (Groombridge, Kabraji & Rao 1987, Groombridge 1989). Local conservationists had been working at nesting beaches near Karachi, in Sind Province (eg. Kabraji & Firdous 1984) but the importance of the Balochistan coast was virtually unknown.

The ground survey confirmed that important nesting occurs at Jiwani and Ormara, of regional and probably global significance. A sample count of nest pits at Ormara in 1987 suggested a minimum of 2,000 green turtle (*Chelonia mydas*) females may have nested that year (Figure 1). The aerial survey reconfirmed nesting in these areas, but found evidence for only sparse nesting at a few additional sites, with no positive evidence for most of the Baluchistan coast.

Government and NGOs in Pakistan have maintained concern for these areas. The Ministry of Environment includes the Balochistan coast in its Wetlands Programme; both Jiwani and Ormara are

now internationally recognised Ramsar Sites, and in partnership with WWF-Pakistan, field activity at Balochistan turtle beaches has recently started (Waqas 2008; Rees 2009). Nevertheless, no substantive new information on Jiwani and Ormara has been published during the past two decades, and apparently no attempt to assess nesting effort more thoroughly has been made. Given the evidence for mass harvesting of nesting females at times during the past three decades (summarised in Groombridge et al. 1987), concern over the status of these populations seems justified.

Recently, while using Google Earth to view the coastline of Balochistan (and recalling the pleasure of travelling there), I was surprised to see that the remote sensing imagery includes unmistakable evidence that dense nesting still occurs on beaches around the western sector of Ormara West Bay.

The evidence is not apparent until the user zooms in very closely, so that individual buildings in the beach villages and individual fishing boats drawn up on the beach are clear and reasonably large, at which point nesting pits can be seen densely scattered on the beach (Figure 2). There are four main beaches separated by areas of less dense nesting (see Table 1; the coordinates provided relate to the approximate midpoint of each beach and can be copied into the Google Earth search box).

The density of nest pits in the latest image (dated January 2004) is very similar to that seen during our ground survey (undertaken in January 1987). Nest pits are also visible, but fewer and less clearly, in images dated September 2003, and are present but still less clear in images dated March 2003. Although no firm numerical data are available it seems reasonable to suspect that the green turtle nesting population using Ormara in 2004 was a similar order of magnitude to that present in 1987 and 1989.

Encouraged by the success of Google Earth as a remote survey tool in this particular case, the other main Balochistan beaches, at Jiwani, were examined. Unfortunately, no definite signs of nesting can be seen in the Jiwani images (dated May 2006, September 2004 and June 2003). Possibly there had been no nesting sufficiently close to the date the imagery was captured, also the resolution of these images appears lower than the Ormara coverage. One short beach area on the southwest edge of the Jiwani peninsula, where nest pits were densely scattered during the September 1988 aerial survey



Figure 1. Green turtle nest pits (and skeletal debris) on one of the nest beaches near Ormara, with view across Ormara West Bay to Ormara peninsula in background. Fresh emergence track on right of beach. January 1987. Author's photo.



Figure 2. Google Earth image of nest beach adjacent to that in Figure 1, near Tak village. January 2004. Horizontal white line represents 100m.

(not visited in 1987) could not be located in the available images. It is not clear whether this is because of tidal conditions, changes in coastal morphology (eg. cliff collapse) or low image quality.

Leaving aside the better known nest sites around the Indian Ocean, large but otherwise unquantified green turtle populations were reliably reported to nest on three beaches along the Hadramout coast of Yemen (Hirth 1968). Similar reports, although lacking adequate documentation, referred to the east coast of Somalia (Travis 1967). Until very recently no later information had emerged about the state of any of these populations. Some attention is now being given to turtles in Yemen (Moqbil 2007), and one of the sites highlighted by Hirth, at Ras Sharma, has nominally been given protection (Stanton 2008).

Beach	Length (kms)	Latitude, Longitude
Ormara 1	0.3	25°16'7.59"N, 64°30'11.07"E
Ormara 2	0.17	25°16'3.36"N, 64°29'59.22"E
Ormara 3	0.22	25°15'53.13"N, 64°29'31.62"E
Ormara 4	1.25	25°15'47.22"N, 64°29'1.22"E
Ras Sharma N	0.43	14°49'31.86"N, 50° 1'32.81"E
Ras Sharma S	0.33	14°49'16.82"N, 50° 1'30.10"E
Jethmun	4.4	14°50'22.75"N, 50° 4'59.14"E

Table 1. Coordinates of main nesting beaches at Ormara, Pakistan and east Yemen (coordinates approximately mid-way along beach).

Examination of imagery in Google Earth does show clear evidence of turtle nest pits on beaches north and south of the Ras Sharma headland (coordinates in Table 1). Nest pits are visible in imagery from June 2009 but are more numerous and densely distributed on the northern beach in imagery captured in November 2003 (not available for the southern beach). I have so far only been able to locate with certainty one other beach listed by Hirth, at Jethmun (= Ithmun). Here the nesting beach is significantly longer, with nest pits along about four km. Again, nest pits are clearest and most densely distributed in November 2003 imagery (present in June 2009 and quite obscure in April 2005). I have not been able to confirm important nesting on beaches in the vicinity of Ash Shuhair, or at Ash Shihr, as

reported by Hirth. Because the imagery is dated July and August, outside the main nesting period, it would be premature to conclude that these populations have disappeared; however, the beaches are not mentioned by Moqbil (2007). I have not been able precisely to locate the beaches in eastern Somalia said by Travis (1967) to support dense nesting, and a brief examination of parts of the coast in Google Earth has so far shown only questionable evidence for sparse nesting.

The utility of Google Earth as a tool to visualise satellite telemetry data derived from migratory animals is now familiar. I note here the fact that Google Earth can also, in favourable circumstances, be used to visualise nesting beaches where heavy green turtle nesting has occurred, and so remotely confirm the recent persistence of a particular nesting contingent. Several conditions must be met for this potential to be more widely fulfilled. It is probably only suitable for green turtles, which tend to leave large and distinct pits in the beach sand. Relatively high resolution imagery is needed, captured during periods when the sunlight is suitably angled and preferably during the nesting season or soon after. The post-nesting period during which nest pits remain visible will probably vary between sites, depending on beach morphology, tidal range, wind conditions, cliff stability, and perhaps human activity.

Perhaps the most important application of this approach will be to stimulate and enable interest among children, students, and others of those countries that have significant green turtle populations and adequately wide access to the Internet. Secondly, it has the potential to confirm the presence of nesting turtles, and hint at the numbers involved, in situations where important nesting has been known or suspected in past decades but where little or no field research has been undertaken during that period. The Makran coast of Pakistan and eastern Yemen, discussed above, exemplify such a situation. Somalia, where anecdotal reports of major nesting exist, may repay further study in Google Earth. If new imagery captured at the most appropriate time and resolution could be accessed on demand, this application could offer a practical survey and monitoring tool for natural resource managers and planners at national level.

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Hawksbill Turtles in Peruvian Coastal Fisheries

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In the Eastern Pacific Ocean (EP), the hawksbill turtle *Eretmochelys imbricata* has been reported as once 'common' from Mexico to Ecuador, but today is extremely rare (Seminoff *et al.* 2003; Gaos *et al.* 2006; Mortimer & Donnelly 2008). In the southeast Pacific, this species has been reported but is not common (Hays-Brown & Brown 1982; Aranda & Chandler 1989; Pitman 1990) and has been documented using the region for both nesting (Green & Ortiz-Crespo 1982) and foraging (Hays-Brown & Brown 1982). In Peru,

hawksbill turtles have been reported from Punta Malpelo (3°30'S, 80°27'W) in the north (Carillo 1987) to Pisco (13°44'S, 76°14'W) in the south (Aranda & Chandler 1989) (Figure 1). Since the latter, there have been no subsequent reports of the species in Pisco (ACOREMA 2000; this study). There are no reports of the species in Chile (Frazier & Salas 1982; Chandler 1991), thus Peruvian waters appear to demarcate the southernmost limit of distribution for hawksbill turtles in the EP.

Month	Season	Year	Location	Gillnet type	CCL	SCL
January	Summer	2000	Mancora	Monofilament	ND	ND
January	Summer	2000	Mancora	Monofilament	ND	ND
September	Spring	2000	Mancora	Monofilament	41.5	38.1
September	Spring	2000	Mancora	Monofilament	35.8	34.2
September	Spring	2000	Mancora	Monofilament	36.5	34.5
September	Spring	2000	Mancora	Monofilament	35	33.1
September	Spring	2000	Mancora	Monofilament	37.9	36.1
December	Summer	2000	Mancora	Multifilament	36	ND
May	Autumn	2002	Parachique	Multifilament	ND	ND
July	Winter	2002	Mancora	Monofilament	47	ND
September	Spring	2002	Mancora	Monofilament	49	ND
March	Summer	2003	Parachique	Multifilament	ND	ND
March	Summer	2003	Parachique	Multifilament	ND	ND
August	Winter	2003	Mancora	Multifilament	28.3	ND
October	Spring	2003	Mancora	Multifilament	ND	ND
November	Spring	2005	Parachique	Multifilament	ND	ND
February	Summer	2008	Constante	Multifilament	37.2	35.2
March	Summer	2009	Mancora	Multifilament	43.7	ND

Table 1. Data from incidentally caught hawksbill turtles in Peru from 2000-09. ND = no data. Curved and straight carapace lengths (CCL and SCL) in cm.



Figure 1. Locations of hawksbill turtle records in Peru from this and previous studies.

Given the scarcity of current data on the hawksbill turtle in the EP (Gaos *et al.* 2010), here we provide a summary of information on the species for Peru, primarily as a result of interactions with artisanal fisheries. Considering recent calls for better information on hawksbills in the EP (CPPS 2008), the information presented herein on the distribution of this critically endangered species will help inform recovery efforts in this region.

Information on hawksbills and their interactions with artisanal fisheries was collected using shore-based observers at eight fishing ports (N→S): Mancora, Constante, Parachique, San Jose, Salaverry, Callao, Pisco and Ilo (Figure 1). Monitoring was undertaken from July 2000 to November 2005 and was conducted opportunistically during subsequent years (2006-2009). Whenever a turtle was landed, observers recorded the species, date, location of capture, and the fishing gear used (Table 1). Curved carapace length (CCL) from nuchal notch to the end tip of the carapace was recorded

Eighteen hawksbills were observed as incidentally caught by fishers operating out of the three northernmost sites: Mancora (n=13, 04°06'S, 81°04'W), Constante (n=1, 05°45'S, 80°51'W), and Parachique (n=4, 05°44'S, 80°51'W). There were no observations of hawksbills at the southern sites from San Jose to Ilo. The majority of interactions (83%) occurred in spring and summer (Table 1). The CCL for hawksbills ranged from 28.3 to 49 cm (mean=38.9 cm, SD=5.9, n=11). The fishing gear in which hawksbills were caught was coastal gillnets, operating within two nautical miles from shore and generally close to mangrove habitats. Hawksbills were the most infrequently bycaught species, with green (*Chelonia mydas*), olive ridley (*Lepidochelys olivacea*), leatherback (*Dermochelys coriacea*) and loggerhead turtles (*Caretta caretta*) all being observed more often (Alfaro-Shigueto *et al.* 2004, 2007).

Hawksbills of the EP have been the focus of increasing conservation attention. Despite a general presumption that the population is at critically low levels, there is a paucity of information on the distribution and size classes of turtles found throughout the EP. While in-water studies of hawksbill turtles are clearly needed to elucidate the abundance and distribution of hawksbill turtles

in marine habitats of the EP, logistic hurdles and the extreme rarity of hawksbills have limited such efforts. Considering the extensive artisanal fisheries operating in this region, we suggest that substantial amounts of information can be gathered through reports of hawksbill-fisheries interactions. Indeed, the data presented here were gathered through a fisheries observation program, which at present is the only mechanism for collecting data on hawksbill turtles in Peru.

The lengths of individuals of the species found in Peruvian waters (Hays-Brown & Brown 1982; Carrillo 1987; this study) suggest that they are mainly juveniles, although adult individuals have also been reported stranded in northern areas of the country (Forsberg 2008). Considering the few records of hawksbills despite nearly a decade of observations, it is clear that hawksbills are uncommon in Peru. Records of hawksbills in the 1980s indicated that the distribution of the species in the EP extended as far south as Pisco. However, we did not encounter any hawksbills further south than Parachique (approximately 1000 km north of Pisco), which corroborates the findings by ACOREMA (2000) in 1999.

Considering the populations' status as one of the most imperiled in the world (Gaos *et al.* 2010) and that the loss of even a few individuals may represent a significant detriment to recovery efforts, these bycatch data require further consideration. Furthermore, there is an urgent need for an updated assessment of the status of hawksbill turtles throughout the EP to learn more about the species' stock origin, abundance and distribution in the region and thus enable the development and implementation of an effective regional management plan.

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First Announcement: Southeast Regional Sea Turtle Meeting, Jekyll Island, GA (USA) February 1 - 4, 2012

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I am pleased to announce the formation of the Southeast Regional Sea Turtle Network and the very first Southeast Regional Sea Turtle Meeting, to be held on Jekyll Island, GA (USA) February 1 - 4, 2012. As a major nesting and foraging site, with 5 species of marine turtles represented, the southeastern portion of the United States has been lacking a formal way for researchers and students to share their knowledge of sea turtle activities that are occurring in this region. After many years of people saying, "I wish there was a regional meeting", I'm happy to say that now there is one. This groundbreaking and historic meeting will allow researchers, students and volunteers to learn about research going on in neighboring states and hopefully allow people to make connections with others who might be doing similar work or even identify those that might be able to aid in their research. Specifically, the presentations will focus on research and activities that occur both on the beaches and in the waters from Virginia through Texas.

Our vision for this biennial meeting started small, harking back to the days of old where us turtlers could kick back with a beer and mingle with old friends while at the same time making new ones. The scope quickly widened when we realized how much interest there was in a meeting such as this. That being said, it is still our goal to make this an intimate, non-intimidating, but yet informative meeting. And don't worry, there will still be plenty of beer and time for friends.

The meeting will be held on Jekyll Island, Georgia. Many of you have traveled to Jekyll Island for past sea turtle related meetings and have already experienced the splendor of this island. For those of you who haven't, Jekyll is a barrier island off the southern coast of Georgia, only 1 hour from Jacksonville, FL and 1.5 hours from Savannah. It is maintained by the state of Georgia, allowing it to keep its old southern charm. Picture Spanish moss hanging from

the trees, grand historic hotels and homes, the sound of cicadas in the air, lemonade or sweet tea on the porch, and of course pristine beaches. Jekyll is the former playground of the rich and famous including the Rockefellers and the Pulitzers. It is now home to the Georgia Sea Turtle Center, our gracious hosts for the very first Southeast Regional Sea Turtle Meeting. While you're here, you should take the time to explore the sea turtle hospital, take a bike ride around the island, play a round of golf, or even take a chance on the casino boat. There are plenty of fun and exciting things to do on Jekyll Island.

As for the meeting itself, expect 2 days of presentations and 1 day of workshops. Presentations will focus on sea turtle activities occurring between Virginia and Texas, USA and we welcome abstract submissions from both researchers and students. The workshops will feature topics that are pertinent to all who are actively working with sea turtles, no matter your expertise or experience. In addition, they will be led by some of the top researchers in the field and all are from our region. Even if you've been in the field for many years, I'm sure you will find a workshop that will intrigue you. Our Welcome Social will be held on Wednesday night, Feb 1, allowing you to catch up with old friends and make a ton more. A taste of Georgia can be experienced on Thursday, Feb 2 when you enjoy a classic Low-Country Boil dinner right on the beach. And on Friday night, Feb 3, get ready for an exciting night of music and auction items! We have a jam-packed schedule of events. But don't let that deter you from exploring the island and enjoying all that Jekyll has to offer.

Please stay tuned for more information as the opening of registration nears. Our website and registration will be launching in August. In the meantime, if you have any questions, please feel free to email me. I look forward to seeing all of you in Jekyll in February.

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

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The remit of the *Marine Turtle Newsletter (MTN)* is to provide current information on marine turtle research, biology, conservation and status. A wide range of material will be considered for publication including editorials, articles, notes, letters and announcements. The aim of the *MTN* is to provide a forum for the exchange of ideas with a fast turn around to ensure that urgent matters are promptly brought to the attention of turtle biologists and conservationists world-wide. The *MTN* will be published quarterly in January, April, July, and October of each year. Articles, notes and editorials will be peer-reviewed. Announcements may be edited but will be included in the forthcoming issue if submitted prior to the 15th of February, May, August and November respectively. All submissions should be sent to the editors and not the members of the editorial board. A contact address should be given for all authors together with an e-mail or fax number for correspondence regarding the article.

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To ensure a swift turnaround of articles, we ask that, where possible, all submissions be in electronic format either as an attached file in e-mail or on compact disc in Word for Windows or saved as a text file in another word-processing package. Should these formats not be suitable, authors should contact the editors to seek alternative arrangements. If internet access or compatible computer facilities are not available, hard copies of the article can be sent to the editors by mail or fax.

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