The Indian Ocean Turtle Newsletter was initiated to provide a forum for exchange of information on sea turtle biology and conservation, management and education and awareness activities in the Indian subcontinent, Indian Ocean region, and south/southeast Asia. The newsletter also intends to cover related aspects such as coastal zone management, fisheries and marine biology.

The newsletter is distributed free of cost to a network of government and non-government organisations and individuals in the region. All articles are also freely available in PDF and HTML formats on the website. Readers can submit names and addresses of individuals, NGOs, research institutions, schools and colleges, etc. for inclusion in the mailing list.

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**Cover photograph:** Hawksbill turtle hatchling from Kuwait
Photo Courtesy: Alan Rees

IOTN is available online at www.iotn.org
EDITOR’S NOTE

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Sea turtle research, monitoring and conservation efforts were viewed through a very different lens as IOTN32 was prepared for publication in July 2020 in comparison to when IOTN31 was published earlier in the year. The impact of COVID-19 and associated social distancing and travel restrictions varies among countries in the Indian Ocean and Southeast Asia, yet members of the sea turtle community- which includes coastal peoples, volunteers, educators, fishers, other local stakeholders, students, journalists, writers, artists, employees of government and small and large NGOs, conservationists, and researchers- will undoubtedly be affected at levels ranging from personal to social to financial regardless of their home country.

IOTN32 presents a regional perspective on the implications of national lockdowns for sea turtle researchers and conservationists, with news from Malaysia, Bangladesh, India, Pakistan, and the Arabian region. The issue also includes research on leatherback satellite telemetry and sea turtle rehabilitation efforts in South Africa, and novel observations of sea turtle distribution and ecology in Pakistan and the Indian state of Maharashtra. Additional articles review the potential for camera trapping studies and in situ nest protection against predators in our region. A resource of potential interest to readers are the recently released National Light Pollution Guidelines from Australia. Finally, we have a report about IOSEA MoS8.

Our thanks to the contributors and reviewers who helped us compile content for IOTN32. The IOTN team is, so far, still able to meet our goals and we look forward to bringing you Issue 33 in January 2021.

CALL FOR SUBMISSIONS

The Indian Ocean Turtle Newsletter was initiated to provide a forum for the exchange of information on sea turtle biology and conservation, management and education and awareness activities in the Indian subcontinent, Indian Ocean region, and south/southeast Asia. If you would like to submit a research article, project profile, note or announcement for Issue 33 of IOTN, please email material to iotn.editors@gmail.com before 1st November 2020. Guidelines for submission can be found on the last page of this newsletter or at http://www.iotn.org/submission.php.
SPECIAL SECTION ON COVID-19

SEA TURTLE EGG ONLINE SHOPPING DURING COVID-19

MOVEMENT CONTROL ORDER (MCO) IN MALAYSIA

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The Phase-1 Movement Control Order (MCO) in Malaysia began on 18th March 2020. The MCO is a cordon sanitaire implemented as a preventive measure by the Malaysian government in response to the COVID-19 pandemic in the country and is commonly referred to in local and international media as a 'partial lockdown'. It prohibits movement and mass assembly of people nationwide, however, essential industries such as supermarkets, small wet markets, grocery stores and multi-functional stores selling daily necessities are allowed to continue their business.

There was a viral issue regarding the increase in sales of sea turtle eggs on social media and internet shopping platforms during the MCO. Among Asian countries, Malaysia consumes the highest number of turtle eggs (Gomez & Krishnasamy, 2019). Eggs are usually sold openly at public markets in the states of Kelantan and Terengganu because the sale is not illegal in these states. However, the closure of large wet markets and restrictions on interstate travel during the MCO prevented such sales. Once sea turtle nesting activity in Terengganu state increased in early April (Chan, 2013), sea turtle egg sellers began accumulating stockpiles of product and sought alternative approaches to reach customers and find new markets. Some sellers used online social media platforms like Facebook, entering social media groups and posting messages to find local customers which allow quick physical deliveries to a designated location for cash-on-delivery. Sellers also moved to online e-commerce platforms, such as Shopee (shopee.com.my), to reach a wider customer base. Courier agencies support such deliveries. A quick search of buyer’s product reviews on the sellers’ accounts reveals that customers received their orders in good (and edible) condition. This model of egg sales is problematic as e-commerce platforms are becoming very popular in Malaysia. As reported by Jabatan Perangkaan Malaysia (2020), more than 90% of households in Malaysia have internet access and the percentage of internet usage in Malaysia increased from 81.2% in 2018 to 84.2% in 2019. The latest official statistics described millions of people as being “glued” to their devices during the MCO. Therefore, the availability of sea turtle eggs on online platforms is likely to increase the demand for eggs and place increased pressure on the conservation of endangered green turtles.

Comments on social media about the viral online sale of sea turtle eggs on Shopee suggest ~10% of people object the idea of a total ban, 20–30% of the public are asking questions that suggest their reactions are neutral, and >50% do not support the sale of turtle eggs. Groups of people supporting egg sellers on social media have argued that egg sellers follow existing regulations and that the law in Terengganu state (Turtle Enactment 1951) allows them to sell any kind of turtle eggs other than leatherback (Dermochelys coriacea) turtle eggs (Jamalludin & Mohd Jani, 2017). Therefore, sellers do not feel that they are acting illegally, and they have never been questioned, arrested, or fined. In addition, there is no mechanism currently in place by which the government can identify whether eggs in the marketplace (physical or online) were sourced from permitted collection sites or protected sea turtle sanctuaries.

Need for Total Ban

The online sale of sea turtle eggs must be addressed as this activity has now crossed the Terengganu border into states such as Pahang and Kuala Lumpur, in which the sale of sea turtle eggs is prohibited (Fisheries Enactment 1937 in Pahang and Fisheries Act 1985 in Kuala Lumpur). Although the sale of sea turtle eggs, other than those of the leatherback turtle, is not prohibited in Terengganu state, we believe the authorities should also prohibit the sales and transportation of eggs into other states. Much research has been done over the past decades to support this suggestion (Cooper et al., 2002; Venkatachalam, 2004; Chan, 2006; Nabangchang et al., 2008; Pattanayak et al., 2008; Jamalludin & Mohd Jani, 2017; Azlina et al., 2019; Mohd Jani et al., 2020) and it is clear that
the sea turtles of Malaysia are conservation dependant. However, the Terengganu Turtle Enactment 1951 is yet to be amended. We believe there is an urgent need for revision and improvement of the current legal situation, and suggest authorities implement a total ban of selling and exporting all species of sea turtle eggs in Terengganu.

It is clearly time to speak out and resolve this issue by seeking the support of a wide range of people and corporations. As a start, members of the Society for Conservation Biology-Malaysia Chapter (SCB-MY) used social media platforms to raise public awareness of the online sale of turtle eggs. SCB-MY is a chapter of an international organisation for conservation professionals dedicated to enhancing the visibility of conservation efforts in Malaysia while encouraging the participation of the public. When SCB-MY members were alerted to the trader selling green sea turtle eggs on Shopee they immediately sprung to action. Members, who include individuals and representatives of government agencies and non-governmental conservation organisations, informed the management of Shopee. As a result, the “product” listing was almost immediately removed, the seller was blocked, and turtle eggs were banned on their platform. Shopee also released a statement that says “Shopee doesn’t tolerate the sale of animal and wildlife products on our platform, as stated in our Prohibited and Restricted Items policy. Users are required to adhere to our policies as well as the local policies, regulations and restrictions set by various governmental agencies and regulatory bodies. We take stern action against users who don’t comply with these standards.” The online commerce platform also asked the public to inform them in the event of any further sale of threatened animals so that they could take appropriate action.

Public concerns about the online sale of turtle eggs on the Shopee.com platform have been shared on social media, and provided leverage for Malaysian sea turtle conservationists to lodge a public complaint to the State Department of Fisheries (DOF). In response to the public complaint, the DOF issued a statement based solely on public advocacy to encourage public to stop eating turtle eggs. The State government released a statement to Agence France-Presse (AFP) (2020) on the 21st May 2020 that the issue would be addressed. The Chairman of Agriculture and Agro-based Industry, Terengganu State Executive Council (EXCO), Dr. Azman Ibrahim, also stated that “Amendments to the Terengganu Turtle Enactment (TTE) will be made”. The question remains whether the state government’s response to this issue is serious and when the amendments or other actions will be made.

As we understand it, legal action has not been previously taken against those who may have illegally collected and sold sea turtle eggs due to confusion regarding the status of nesting beaches. For example, some nesting beaches in Terengganu state are not gazetted as sanctuaries, and landowners on these non-sanctuary beaches can bid for license from the State DOF to collect sea turtle eggs. The collected eggs can be sold but must be offered to the State DOF first. If sold to the State DOF, the eggs are transferred to hatcheries for incubation. If the State DOF chooses not to buy eggs, then sellers have the right to sell eggs in public. This tender system has previously provided necessary funding to the State government to operate their sea turtle conservation efforts. But, turtle eggs are often advertised as originating from Redang Island, which is in a Marine Park from where it is illegal to collect and subsequently sell eggs. Were these eggs really collected from Redang Island for illegal sale? Or were they legally collected from elsewhere and the source beach mistakenly named?

We strongly recommend that the Government of Malaysia take action such as apprehending people collecting eggs illegally. Egg sellers often say that eggs received are from captive breeders, but this is not likely to be a true statement as no turtle farms have been established nationwide. Documentation that eggs were offered to the State DOF before sale to the public should be made available, and the sellers should also accurately identify the beach that eggs were collected from.

**Political Will**

The current political scene in Malaysia poses another challenge to sea turtle conservation. Political turmoil frequently affects the initiatives of the State DOF. However, in 2019, legislature took seriously the issue of conserving Terengganu’s turtles when they invited scientists to join the state legislation process. There is no political will by politicians who reject addressing the issue because they fear losing local supporters. Failure to take this opportunity is likely to be detrimental to Malaysia’s turtles. Malaysians are very aware of environmental issues due to nationwide campaigns that have been ongoing for more than 10 years and we believe the momentum of public opinion is already there. Politicians should take this opportunity to amend the TTE to support a total ban on the sale of turtle eggs throughout Malaysia and make law enforcement easier. A response on this issue will enhance their credibility as a progressive government.

The Terengganu State’s current government should also re-evaluate the practice of tendering licenses to egg collectors. Egg collectors can experience a financial loss if the number of eggs that are collected and sold within a season does not result in a profit greater than the cost of the lease. Islamic finance jurisprudence does not permit such an uncertain element (known as *gharar* in Arabic) that might harm any party, so there are grounds for the
State Government to address the *halal* (lawful) lease status.

**Next Steps**

Turtles are an iconic species in Terengganu, with multiple conservation projects striving to ensure future populations. Currently, the state has the highest number of recorded landings for green turtles (*Chelonia mydas*) within Peninsular Malaysia (Chan, 2006), but many turtle eggs sold at the market. In the 1970s-80s, large numbers of nesting leatherback turtles attracted foreign tourists with economic benefits for local economies. However, by the early 1990s, the pressures of egg collection and fisheries bycatch had drastically reduced the number of leatherback turtles landing and the iconic species is now considered to be locally extinct (Chan *et al.*, 1988; Chan & Liew, 1996; Liew, 2011; Mohd Jani *et al.*, 2020). A downturn in tourism and local income was attributed to loss of this valuable natural asset (Chan & Liew, 1996; Ibrahim & Sharma, 2006; Liew, 2011) so there are socioeconomic as well as ecological reasons to conserve remaining turtle populations.

In conjunction with World Sea Turtle Day on 16th June 2020, SCB-MY organised public awareness programs including a two hour discussion about the challenges and opportunities of sea turtle legislation in Malaysia. We also initiated a public pledge to “Say No to Turtle Eggs” on change.org and received 8,500 pledges within a week. The majority of Malaysian Muslims will take their pledge seriously as it is regarded as a promise they must keep.

Malaysians eat turtle eggs but not turtles. Malaysians are a Muslim majority country and follow the clerics’ explanation obtained by madhab Shafie (one of the four great Imams) that all eggs are *halal* (permissible in Muslim dietary law) because the nature of the egg itself has not yet been formed (embryonic development). An exception is eggs proven to be harmful. However, turtles are animals that live both on land and in water, so their consumption is *haram* (prohibited).

We were recently approached by Islamic centre officials to further discuss the implications of eating turtle eggs for the environment and human health. The declining trends of some turtle populations and their important roles in the ecosystem (Bjorndal & Jackson, 2002) underpin the need for their conservation, but turtle eggs also have high cholesterol content and potentially contain heavy metals (Kaska & Furness, 2001; Aguirre *et al.*, 2006; Lam *et al.*, 2006; Merwe *et al.*, 2009; WWF-Malaysia, 2009; Joseph *et al.*, 2014) so their consumption can result in health conditions. Such scientific evidence can be used to produce a *fatwa*, a formal interpretation on a point of Islamic law made by a qualified legal scholar. Malaysians have a high religious compliance so this new action of producing a *fatwa* to forbid eating turtle eggs in Islamic law will likely be effective.

More environmental education and initiatives to improve ocean literacy are also required, especially among Malaysian youth who are disconnected from nature (Nathan *et al.*, 2019). We encourage parents and teachers to use the Phase-Recovery MCO (after 10th June, 2020) as an opportunity to take the classroom to the beach, where social distancing can still be practiced, and learn about the importance of conserving our natural environment and engage with sea turtle conservation. Many NGOs have also created online classrooms and virtual field trips to raise awareness of environmental issues. Engaging all demographics of the community with work of NGOs and government agencies is needed to conserve sea turtles.

**Literature cited:**


**IMPACT OF COVID-19 ON SEA TURTLE NESTING, CONSERVATION AND MANAGEMENT IN BANGLADESH**

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In Bangladesh, the major sea turtle nesting season is in winter (October to March/April). The national lockdown in response to the COVID-19 pandemic began on 26th March 2020 and has, to date, continued throughout June. Within the lockdown period, human activity continued as many people are daily wage earners or operate small businesses, especially in major cities. In coastal areas, the lockdown was more strictly followed and there was no tourism. Some fishing activities continued, although it was not supposed to during the lockdown.

The NGO Marinelife Alliance (MLA) runs a community-based research and conservation program involving local conservation assistants (CAs). All 56 CAs live in beach front villages along 350km of the southeast and south-central coast. They continued conducting night
observation and mitigating threats such as disturbance of nesting turtles and illegal take of eggs; this task was easier than usual during the lockdown because there was no crowding and disturbance. A setback was that biologists and central researchers could not move to the field. This challenge was overcome by the well-trained CAs sharing their data via cellphone and social media. Another major impact of the lockdown was that ecotour operators who previously provided financial support for MLA conservation activities could not do so. The lack of tourists has allowed beach vegetation and invertebrates to flourish during the lockdown, with sand dunes forming along the beach and beach morning glory (Ipomeas pp.) growing to protect the shoreline and support beach biodiversity.

COVID-19, CYCLONES AND SEA TURTLES IN INDIA

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The COVID-19 pandemic resulted in a sudden nationwide lockdown in India on 24th March 2020. The nesting season of leatherback and olive ridley turtles start from November and lasts until April both on the Indian mainland coast and Island territories, while green and hawksbills nest throughout the year. The nationwide lockdown did not have any significant influence on sea turtle nesting, monitoring and conservation activities.

On the east coast, olive ridley turtles usually nest from December to March, and hence most of the nesting had concluded when the national lockdown was announced. In the state of Odisha, mass nesting (arribada) took place at Rushikulya rookery from the 21st to 26th March 2020, including a day-time nesting event, which partially overlapped with commencement of the lockdown. While it is not uncommon to have turtles nesting during the day in these arribadas, some of the Indian media channels reported this phenomenon to be a result of reduced human activity on the beach (e.g., Das, 2020; Express News Service, 2020; Gill, 2020).

Researchers from Dakshin Foundation continued to monitor the beach till the 24th March 2020. It was estimated that over 200,000 turtles had nested during the first three days of the arribada. Despite the lockdown, the Odisha Forest Department personnel were present for the mass hatching to ensure that disoriented hatchlings were released at the water’s edge. During the lockdown, there was a cyclonic event (Cyclone Amphan) on the 16th May 2020 that was expected to affect the Odisha coast; however, the cyclone moved north towards West Bengal and resulted in no damage to the critical nesting beaches of Odisha.

In Tamil Nadu, the Students’ Sea Turtle Conservation Network (SSTCN) reported that the nesting season had ended prior to the lockdown, however, the clutches relocated to the hatcheries continued to hatch till May. In collaboration with the Tamil Nadu Forest Department, SSTCN members were granted permission from the local authorities to visit the hatchery and ensure that the hatchlings were released in a timely manner.

The west coast of Maharashtra has low-density nesting. Nevertheless, the village of Velas hosts an annual “Turtle Festival”, a sea turtle-based ecotourism initiative jointly run by the village panchayat (local government) and the Mangrove Foundation of the Maharashtra Forest Department. Since the lockdown led to cancellation of all tourism related activities, the Mangrove Foundation hosted the Turtle Festival on Facebook Live and broadcast hatching releases every morning and evening for viewers to witness from their homes. However, on 1st June, 2020, the Maharashtra coast was hit by Cyclone Nisarga which caused heavy damage to sheds and other structures used to protect the nesting sites in Velas and other sea turtle nesting beaches in Anjarle, Dabhol, Kelshi etc. Fortunately, there was no damage to the beach, and since the cyclone had occurred after the end of the nesting season, monitoring and ecotourism activities were not been hampered.

With most organisations working from home and online, there was an increase in outreach sessions conducted in the form of webinars, especially on the occasion of World Turtle Day (23rd May 2020). The themes of these webinars varied from understanding sea turtle biology and conservation to the work being conducted in the region and the experiences of sea turtle biologists. These sessions were conducted in vernacular and English and were hosted by a range of news media outlets like the
Mumbai MTB, research centres like the Kalinga CRE and government organisations like the Mangrove Foundation.

On the whole, the lockdown did not have a major impact on any monitoring activities since the nesting season across India was nearing its end. Most organisations and individuals, approved by relevant local Government authorities, were able to continue any required monitoring activities without much disruption. With the reduction in fishing activities and other anthropogenic activities on the beaches during the lockdown, the impact of associated threats to sea turtles and their habitats were considerably lessened, though temporarily.

Literature cited:

The COVID-19 pandemic triggered a major lockdown in Pakistan. There have been some positive outcomes for biodiversity and conservation, including local marine turtle populations, from this response. Adjacent to the metropolis city of Karachi in the Sindh Province are two major nesting beaches- Sandspit and Hawksbay- which are usually thronged by large numbers of picnickers. However, the beaches have been deserted since the last week of March 2020 because of the national lockdown. The peak visiting period is usually between March and September, which overlaps with the peak turtle nesting period from July to December (although nesting may occur year-round). Previously, high human presence on these beaches has disturbed nesting turtles by trampling the nests, damaged eggs after opening the nest and leaving it exposed to sun, scavengers and predators, and disrupted the seaward movement of emerged hatchlings by blocking their movement or picking them up and releasing them when the hatchlings are exhausted. The amount of solid waste pollution, which may be an obstacle for nesting turtles and hatchlings, has also reduced during the lockdown. A similar situation has prevailed on other beaches along the Sindh coast and on Astola Island in the Balochistan Province, where security forces did not allow fishing or tourist boats to visit. However, the situation is different elsewhere in the Balochistan Province. The number of people visiting Daran Beach at Jiwani substantially increased during the lockdown, due to low compliance with the order for restricted movement, and could have affected sea turtles and hatchlings as it did in Karachi However, a major portion of Daran beach was extensively eroded by Cyclone Kyarr in late October of 2019, and it has not yet been reestablished by wave and tidal action; no turtles have been reported nesting since November 2019. Therefore, nesting turtles and hatchlings at Daran have not been impacted by the lockdown.

Despite strict control of the Sindh Wildlife Department, a small illegal trade of turtle hatchlings in pet shops, aquaria and the Sunday pet market occurs in Karachi. Hatchlings are removed from the nests along the Karachi and Balochistan coast and are illegally sold in these outlets. Since all markets have been completely closed during the lockdown, no such illegal sales have occurred since the last week of March 2020. The government imposed a ban on fishing as part of the lockdown along the entire coast of Pakistan since
the last week of March till mid May 2020, and this has presumably reduced the threat of entanglement in fishing gear, subsequent injury or mortality, and disturbance to foraging areas. Partial fishing operations (daytime fishing only and within 5 nautical miles of the coastline) recommenced in mid-April. However, the annual two month closed season for shrimp trawling and tuna gillnetting in the Sindh Province along with a total fishing ban in the Balochistan Province will be in place from the 1st June to 31st July, 2020, and hence will provide additional protection for turtle populations in the coastal and offshore waters of Pakistan.

RECENT SEA TURTLE UPDATES FROM THE ARABIAN GULF

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UNITED ARAB EMIRATES

Sea turtle nesting in the UAE usually begins in March and ends in June, so it overlapped with the lockdown period during the COVID-19 pandemic. Emirates Marine Environment Group (EMEG), an NGO supported by public authority, monitors sea turtle nesting at Jabel Ali Reserve (JAR), Sir Bu Na‘air (SBN) Island, and 30 other islands. Two decades of records are available for JAR and SBN, the two core sites of the monitoring project. EMEG reports that higher numbers of hawksbill sea turtle nests at SBN have been recorded this year than in the previous year (EMEG, 2017, 2018, 2019; Figure 1). When 400 nests were reached on 12th June 2020, the head of the Environment and Protected Areas Authority (EPAA), of Sharja, visited SBN Island to celebrate what is believed to be the highest number of annual nests of the species in the Arabian Gulf region. At JAR, 52 hawksbill nests were recorded last season and 45 nests have been counted to date in the 2020 nesting season. Green turtle nesting is always very low, with only one found annually at SBN and none at JAR.

The main impact of COVID-19 on turtle conservation work in the UAE is the reduced number of available field workers at SBN, down from 14 in 2019 to four in 2020 (including the EMEG President who will remain on the island for the entire season). Members of the smaller team worked two of three shifts each day, monitoring nesting sea turtles and seabirds (e.g., sooty gulls and bridled terns) as well as conducting dive missions related to the conservation of coral reefs. Fewer people also meant that beach cleanups were not possible at the start of the nesting season, and waste on the beach might have been an obstacle for nesting turtles (EMEG, 2019). However, EMEG, in collaboration with EPAA, organised a cleanup day for SBN beaches on World Environment Day, 5th June 2020.

QATAR

Hawksbill sea turtles mainly nest on mainland beaches of Qatar, unlike other Arabian Gulf countries where nesting occurs on islands. Qatar closes some, but not all, of its beaches from April until the beginning of August each year. However, the COVID-19 lockdown meant that there was limited use of most beaches from February. The number of recorded hawksbill turtle nests has varied

Figure 1. Number of hawksbill turtle nests recorded at Sir Bu Na‘air Island over time (EMEG, 2017, 2018, 2019).
over time (Figure 2) based on the data from the Qatar Ministry of Municipalities and Environment, the study team at Qatar University (@albaladeya, 2020), isolated records from remote locations, and periodic visits to some locations by independent field researchers. Field researchers informally visited beaches to monitor turtle nesting during the COVID-19 lockdown and reported more nests than in previous years. For example, the number of nests on Fuwairit Beach have increased from 91 in the entire 2019 season to 118 by the beginning of June 2020 (@albaladeya, 2020), with nesting expected to continue until August. Up to mid-June, 380 nests had been counted in the country (Alyafei, pers.obs).

This season, the first hatchlings for Fuwairit Beach were observed on the 10th June (@albaladeya, 2020). The hatching success (proportion of eggs that hatch to produce a hatchling) in Qatar ranged from 70-80% between 2016 to 2019 and there have been no environmental conditions or changes that would cause a decrease in 2020. As the number of nests is expected to increase in 2020, the total number of hatchlings that emerge during the season is also expected to be higher than that for 2016-2019 seasons.

BAHRAIN

Sea turtle nesting has not been recorded in Bahrain, but the Environment Friends Society (EFS) has records of elder fishers recalling nests being laid on remote islands of Bahrain in the 1990s. Since 2013, stranded sea turtles have been rescued and treated by the Bahrain Sea Turtle Rescue Team (BSRT) before their release back into the sea. Fishers, people on the beach, and cleanup campaigns report and deliver turtles requiring treatment to BSRT, often including hatchlings stranded with beach debris.

December till April is usually busy with turtle rescues, treatment, and rehabilitation for release. By April 2019, EFS and BSRT had received 10 hatching hawksbills, two juvenile hawksbills, and two adult green turtles. Monitoring of social media has also shown turtles being found by fishers or other members of the public, cleaned of barnacles, and released during these months. During the COVID-19 lockdown, fewer fishers have been out at sea and beach cleanups are non-occurring. Hence, there have been fewer opportunities for sea turtles to be found and delivered to EFS or BSRT. So far in 2020, BSRT has received no turtles but social media posts show ~10 turtles found entangled in ghost gear and released by fishers, while in the area or observed floating at sea (Al-Muhannadi, pers. obs).

SUMMARY

More nests have been or are expected to be laid in the 2020 nesting season compared to the previous years in the UAE and Qatar. There has been a decrease in disturbance on nesting beaches due to the COVID-19 lockdown. However, the lockdown also decreased the number of field workers available for monitoring sea turtles and hence, there may not be records of all the nests. This also meant that there was a decrease in the usual rescue and treatment of stranded turtles. There are also fewer beach cleanup activities, which can result in beach debris accumulation and make it challenging for nesting turtles and hatchlings to navigate the beach to the sea.

Literature cited:


Figure 2. The number of hawksbill sea turtle nests in Qatar from 2007 to 2019.
INTRODUCTION

Long-distance, ocean-traversing migrations are well documented for leatherback (Dermochelys coriacea) sea turtles (e.g., Luschi et al., 2003, Benson et al., 2011), and are also a paramount driver of leatherback conservation challenges; over such expansive distances, turtles are exposed to many threats in multiple jurisdictions and on the high seas (Wallace et al., 2011). Many populations are still in peril (Wallace et al., 2011), with indications that key pressures limiting population recovery are often offshore (Harris et al., 2018). Therefore, beach protection alone is not a guarantee for conservation success (Nel et al., 2013). Marine Protected Areas (MPAs) can be effective in protecting turtles in areas where they aggregate, like feeding or courtship areas. This provides strong motivation for considering migration and foraging distributions of leatherback and other turtles as countries embark on Marine Spatial Planning (MSP) initiatives or MPA expansion programs. These could be used in concert with other conservation measures, particularly in the South Western Indian Ocean (SWIO) where leatherbacks are Critically Endangered (Wallace et al., 2013).

The protection of migratory species is especially complex in developing nations, which face pressing needs to use natural assets for food security, job creation and security, poverty alleviation, and tourism, while conserving natural biodiversity. South Africa, for example, has implemented a National Development Plan which prioritises the expansion of an ocean-based economy. It aims to combine biodiversity protection with sustainable development and expansion of marine livelihoods through Operation Phakisa (www.operationphakisa.gov.za). As part of this framework, South Africa expanded its existing ocean protection with 20 new MPAs (DEA, 2019), which increased the protected proportion of ocean territory around mainland South Africa from 0.4% to 5.4% (Sink et al., 2019). Representative protection of ecosystem types, spectacular features, key life history areas and distributions of focal species such as leatherback turtles, were part of the motivation, design, and zonation of these new MPAs.

Nominations for two of the new MPAs in particular were based on the seasonal presence of leatherback turtles. iSimangaliso Wetland Park is a key nesting site for the SWIO loggerhead (Caretta caretta) and leatherback management units (Nel et al., 2013), with ~120 km of protected nesting beaches and originally 5 km of protected internesting area offshore. The more extensive internesting movements of leatherbacks (Harris et al., 2015) are now protected by a new iSimangaliso MPA extending 186km alongshore, and ~20-33km offshore (DEA, 2019; Figure 1). Another new MPA, the highly productive Agulhas Front MPA, (colloquially called the “Turtle Tuckshop”), was also in part proclaimed as leatherbacks seem to forage in oceanic frontal areas (Harris et al., 2018) with the u’Thukela, shelf edge and seamount–associated MPAs also recognised as potentially important for turtles.

In a complementary initiative, South Africa has been revising its Ecologically or Biologically Significant Marine Area (EBSA) network as part of the Marine Spatial Management and Governance Programme (MARISMA 2014-2020) under Operation Phakisa. EBSAs are spatially discrete areas that perform important ecosystem services and/or host unique or vulnerable biodiversity relative to adjacent marine areas. They were conceptualised by the Convention on Biological Diversity (CBD) and are considered helpful to guide countries’ efforts to achieve
their Aichi biodiversity targets (UNEP-CBD, 2012). To be inscribed as an EBSA, a site must meet at least one of the seven EBSA criteria (UNEP-CBD, 2009), including importance for life-history stages and/or importance for threatened species. South Africa’s revised network includes 23 EBSAs (MARISMA Project, 2020), and for some of these EBSAs, sea turtle presence contributed to the sites meeting these two criteria. EBSAs are not legally binding, but the CBD encourages countries to implement improved conservation and protection measures to secure the special biodiversity features within them. All South Africa’s new MPAs reside partially in EBSAs, and in combination they are key to marine spatial planning (MSP) processes, ensuring the protection of important biodiversity features, including leatherback turtles.

Previous satellite telemetry studies have indicated three migratory paths for post-nesting leatherback females: northerly into the Mozambique Channel, south-westerly into the South Atlantic, and a south-easterly migration into the open waters of the SWIO (Harris et al., 2018; Robinson et al., 2018). Less attention has been given to the “exceptions” that seemed to be aimless “wanderers”. Several leatherback turtles from these previous tracking studies seem to have meandered along the coast, but satellite transmissions terminated before they reached an identified foraging ground or returned for the next season’s nesting.

Sea turtle nesting beaches in South Africa are well protected in the new iSimangaliso MPA, but the leatherback population is still small with 60-70 females nesting per season (Nel et al., 2013). An important task that remains to protect and conserve SWIO leatherbacks is thus to identify foraging areas, especially in relation to existing or future MPAs that may not only provide protection to turtles but also sufficient productivity and diversity to sustain their foraging needs. It would also be relevant to ascertain the mode by which leatherbacks benefit from MPAs, as their gelatinous diet seem to have unidentified origins. Jellyfish distributions shift interannually and seasonally but seem to attain highest densities in shallow water (<200m), with high zooplankton abundances, and proximity of hard substrate for polyps to attach (Flynn et al., 2011). It is, therefore, possible to identify spatial areas with high current or topographic advection driving local productivity and consequently jellyfish abundance. If these conditions are combined with a network of sites with reduced threats because of, for example, displaced fisheries, leatherback turtles may benefit.

The aim of this paper is, thus, to inspect the distribution of leatherback turtles from the SWIO and evaluate the conservation role of the new South African MPAs and EBSAs in protecting leatherback turtles.

**Satellite Deployment and Tissue Sampling**

Three leatherback turtles that nested on the beaches within iSimangaliso MPA were fitted with Sirtrack/LOTEK FastGPS/Argos satellite tags (model F6H 473A) between 12-15th January 2020. One turtle was a neophyte (or previously untagged turtle) and the other two were remigrants that had nested in previous seasons. Turtle Jackie (flipper tag: ZA1471E; satellite tag: 196916) was the neophyte flipper tagged by Jackie Raw in December 2019. Turtle Michaela (named after volunteer, Michaela King) with flipper tag NN145 and satellite tag 196913 nested for the first time in 2012 and nested again in 2020. From tag scars, it was suggested that Turtle Tamsyn (named after MPA GIS expert, Tamsyn Livingstone; flipper tag: ZA1708C; satellite tag: 196915) had also nested previously but in an unknown year due to both flipper tags being lost. All three turtles left the beach with a satellite tag directly attached to the dorsal ridge, and two metal flipper tags applied to the trailing edge of the rear flipper.

Epibionts and body shape are proxies of body condition whereas blood metabolomics provide a clinical indication of health (Nolte et al., 2020). Size was, therefore, measured, and blood samples were collected from the femoral retae system in the rear flippers (following Dutton, 1996) using a 20-gauge needle and heparin-coated vacutainers that were kept on ice and processed for packed cell volume (PCV) and total solids (TS) at the field station following Perrault et al. (2012) as early indicators of health.

**Results**

Tagging of leatherback turtles in 2020 had mixed success: two of the three satellite tags failed within 10 days while the third tag transmitted until 2nd July 2020 (Figure 1). Published tracks (Figure 2) from the same rookery from satellite tags deployed in the nesting seasons of 2011-13 (Robinson et al., 2016), and between 2006-2014 (Harris et al., 2018) were, therefore, added for further comparisons.

The satellite tag deployment (196915) on Tamsyn was active for 170 days. During that time, she made a southward journey in the Agulhas Current but, unlike most of her predecessors (e.g., Figure 2) that tended to move directionally south-westwards with the current, Tamsyn made several “small” clockwise loops (Figure 1), each lasting 10-20 days. During these excursions she swam south with the current and returned along the coast in a northerly direction. The only exception was one anticlockwise rotation where she left the shelf off East London, and then returned back to the coast further north. Her last transmission, after visiting the southernmost MPA (#24; South Western Indian Seamounts), was back up to East London to the Amathole (#32) and Amathole Offshore (#33) MPAs and the Algoa to
Figure 1. Satellite track of Tamsyn the turtle (black line) between 15th January and 19th June 2020 relative to all MPAs and EBSAs in South Africa’s mainland marine territory, with an enlarged insert showing the tracks of Michaela and Jackie near the nesting beaches in iSimangaliso. The -500m isobath is plotted as a dashed grey line to show the split between the shelf edge and slope. For more information on South Africa’s MPAs see: www.marineprotectedareas.org.za, and for more information on the EBSAs see: https://cmr.mandela.ac.za/EBSA-Portal/South-Africa. Note that some MPAs comprise more than one area, e.g., 24, 32 and 33. For coloured areas, see the pdf version, available on-line.
Amathole EBSA. The total longshore coastal distance covered until 19th June was ~1,400km but the track length was seven times longer with a swimming distance of 10,069km. The satellite tag deployment (196915) on Tamsyn was active for 170 days. During that time, she made a southward journey in the Agulhas Current but, unlike most of her predecessors (e.g., Figure 2) that tended to move directionally south-westwards with the current, Tamsyn made several “small” clockwise loops (Figure 1), each lasting 10-20 days. During these excursions she swam south with the current and returned along the coast in a northerly direction. The only exception was one anticlockwise rotation where she left the shelf off East London, and then returned back to the coast further north. Her last transmission, after visiting the southern-most MPA (#24; South Western Indian Seamounts), was back up to East London to the Amathole (#32) and Amathole Offshore (#33) MPAs and the Algoa to Amathole EBSA. The total longshore coastal distance covered until 19th June was ~1,400km but the track length was seven times longer with a swimming distance of 10,069km.

While Tamsyn travelled along the east coast of the country and offshore to the shelf edge and waters overlying the slope and sea mounts south of St Francis Bay, she traversed ten coastal and two offshore MPAs. The journey started in iSimangaliso MPA (#41) with GPS transmissions received from every coastal MPA except Dwesa-Cwebe (#34; although there were transmissions on either side and hence it was inferred that she travelled through), Sardinia Bay (#30) and Tsitsikamma (#27; Figure 1). She also visited every EBSA east of Mossel Bay (i.e. north to south: Delagoa Shelf Edge, Canyons and Slope; KwaZulu-Natal Bight and uThukela River; Protea Banks and Sardine Route; Algoa to Amathole; Kingklip Corals; and Shackleton Seamount Complex) except Tsitsikamma-Robberg along the south coast. Her track also confirms high use of the shelf, shelf edge and slope waters along the eastern seaboard of South Africa (Figure 1).

At this stage we do not know the final foraging destination of these three sea turtles, as the three tags failed before they started the return journeys for the next nesting season. However, the proxy indicators of condition and
health suggest that all three animals (irrespective of being a coastal or pelagic feeders) were in good condition with carapace lengths exceeding 160cm. The neophyte turtle (Jackie) was the longest of the three, but the girth of Tamsyn (and so body condition) was substantially more than the other two turtles. The blood metrics, particularly packed cell volume supported the good condition for Tamsyn with a value of 43%, which is ‘normal’ although slightly better than the other two females (Table 1). She laid a clutch of 100 eggs at the time of tagging (which is the mean for the iSimangaliso population), whereas the other two females had slightly larger clutches (Table 1).

**DISCUSSION**

The aim of this paper was to review the post-nesting distribution of leatherback turtles from iSimangaliso and to reassess likely migratory routes between feeding and nesting grounds. Although the sample size of this study is small, it provides a valuable comparison with previous tagging studies (and the first blood metrics), thus adds to a baseline dataset to measure future performance of these MPAs and EBSAs for leatherback turtles. The conservation measures for the SWIO leatherbacks to date have been centred around long-term beach protection exceeding 55 years, but despite these efforts, the population has remained Critically Endangered (Nel et al., 2013; Wallace et al., 2013). The MPAs and spatial protection measures intended to be implemented in the EBSAs through the MSP process may be a “game changer” for this species.

Our results suggest that the location of the areas with spatial protection (MPAs and new measures proposed within EBSAs) are better suited for sea turtles than previously recognised. The new MPA not visited by Tamsyn ironically is the Agulhas Front “Turtle Tuckshop”. However, the other two published studies indicated the popularity of this MPA with post-nesting females with a number of tracks circling through the Agulhas Front, probably *en route* to the Atlantic or SWIO foraging grounds, particularly those turtles migrating further offshore (Figure 2).

The unusual migration path of Tamsyn is not due to poor body condition or disease; she is an average-sized individual for South African leatherbacks, which range from 159.3-196.3cm for neophytes and remigrants (Le Gouvello et al., 2020), with apparently normal blood metrics. These are the first baseline PCV and TS data for South Africa, which are comparable with published studies from Florida and Georgia (Perrault et al., 2012), Gabon (Deem et al., 2006) and Bioko (Honavar et al., 2011). PCV for nesting females in Florida (n=59) were (mean±SD) (range) 38±4.4% (27-50) and direct captures for Georgia (n=18) 42±7% (24-49), with Gabon (n=28) 36±5.4% (28-56) and Bioko (n=55) at 36.4±5.09% (27.8-44.9). TS (g/Dl) for Gabon (mean±SD) were 4.0±0.7 (range 2,3-5.4) and Bioko (n=54) at 5.08±0.1 (3.6-6.56). The PCV for Tamsyn was thus on the higher end of the mean range for nesting turtles, whereas the other two were on the low end but within one SD. Jackie, the neophyte, had a low TS value with the remigrants both in the mean range of nesting females from the Pacific (Harris et al., 2011). There is still much to learn from the metabolomics, but the combination of foraging location and nesting condition suggest that the eastern seaboard is an adequate foraging ground if leatherbacks do indeed remain along this coast.

It is generally assumed that post-nesting females return to the same foraging grounds from which they come (Marcovaldi et al., 2010), which suggests that turtles with similar distributions as Tamsyn may spend a substantial amount of time along the eastern seaboard. These MPAs, therefore, seem to be attractive to some migrating leatherbacks. The mode of attraction is intriguing and not entirely clear, as the preferred food of leatherbacks are planktonic drifters and thus there seems little dependence on the (spatially predictable) benthic biodiversity as

**Table 1. Tagging information, blood results, and body condition of tracked turtles. CCL- curved carapace length; CCW- curved carapace width; PCV- packed cell volume; TS- total solids. Neophytes- first time observed to nest; Remigrants- tagged or with tag scars.**

<table>
<thead>
<tr>
<th>Turtle</th>
<th>Satellite Tag</th>
<th>Flipper tag</th>
<th>Date</th>
<th>Distance*</th>
<th>Nesting status</th>
<th>CCL</th>
<th>CCW</th>
<th>Eggs*</th>
<th>Blood PCV % : TS g/dl</th>
</tr>
</thead>
<tbody>
<tr>
<td>Michaela</td>
<td>196913</td>
<td>NN145</td>
<td>15 Jan 2020</td>
<td>69 km</td>
<td>Remigrant</td>
<td>161.2</td>
<td>115.0</td>
<td>136</td>
<td>32 : 4.2</td>
</tr>
<tr>
<td>Tamsyn</td>
<td>196915</td>
<td>ZA1708C</td>
<td>12 Jan 2020</td>
<td>43 km</td>
<td>Remigrant</td>
<td>164.4</td>
<td>122.0</td>
<td>100</td>
<td>43 : 4.3</td>
</tr>
<tr>
<td>Jackie</td>
<td>196916</td>
<td>ZA1471E</td>
<td>15 Jan 2020</td>
<td>51 km</td>
<td>Neophyte</td>
<td>165.4</td>
<td>113.2</td>
<td>124</td>
<td>31 : 3.2</td>
</tr>
</tbody>
</table>
other turtle species would. However, highly productive shelf, shelf-edge and deeper waters, now encompassed in MPAs and EBSAs, may include productive upwelling zones that seem to provide at least an interim destination (or half-way stop) for post-nesting females to replenish. This south-eastern edge of the African continent is characterised by the edge of the Agulhas Bank forcing the Agulhas Current offshore, and a combination of log-spiral bays and rocky headlands with strong summer north-easterly winds. The result is wind- and current-driven coastal upwelling (Goshen et al., 2012) and potential retention of zooplankton accumulations along the coast and in bays. These oceanographic features are similar to those described in Benson et al. (2011) including current boundaries, stationary fronts and coastal retention areas aggregating prey, and so attracting Pacific leatherbacks to coastal foraging grounds.

The residence duration of leatherbacks on the Agulhas shelf seems to differ among the studies (Figure 2) suggesting interannual differences in behaviour that are most likely dependent on oceanic conditions of any given year, and the foraging destination from which the turtles have come. Both Robinson et al. (2016) and Harris et al. (2018) clearly demonstrated migratory routes to the north, southwest and southeast. Tamsyn, however, seems to be following a different post-nesting migration strategy, remaining well within South Africa’s marine territory. This indicates that there is likely an undocumented fourth post-nesting migration/foraging route (or interim stop-over) for leatherbacks in the SWIO, but detailed analysis of oceanographic and track data is required to confirm the drivers and conditions under which the eastern Agulhas Bank is a preferential foraging destination.

CONCLUSION

Tamsyn the leatherback turtle frequented 12 of 13 MPAs on the South African eastern seaboard. This data set, although preliminary, suggests that South Africa’s expanded MPA network is likely to contribute to the protection of leatherback turtles during nesting, migratory and foraging stages. It is the first time that FastGPS tags were used for South African leatherbacks, providing better spatial resolution than available from traditional Argos tags, especially for data points close to inshore. This improvement in accuracy has contributed to better determining the potential value of the MPAs and the delineation of EBSAs for protecting post-nesting females, with this area also potentially being a distinct foraging ground. It is necessary to review these data periodically because the new MPAs may aid in the recovery of SWIO leatherbacks and other, more coastaly distributed, sea turtles.

ACKNOWLEDGEMENTS

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A REVIEW OF STRANDED MARINE TURTLES TREATED BY USHAKA SEA WORLD (SAAMBR) IN DURBAN, SOUTH AFRICA

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INTRODUCTION

Of the world’s seven sea turtle species, five are found in South African waters (Sink et al., 2019). The southwest Indian Ocean subpopulations of loggerhead (Caretta caretta) and leatherback (Dermochelys coriacea) turtles nest on the beaches of northern KwaZulu-Natal (KZN) while green (Chelonia mydas) and hawksbill (Eretmochelys imbricata) turtles feed and mature in several offshore ecosystems along the South African coast (Sink et al., 2019). Globally, loggerhead and leatherback turtles have been assessed as Vulnerable on the IUCN Red List (Wallace et al., 2013a; Casale & Tucker, 2017), but regional populations in the Southwest Indian Ocean have been assessed as Near Threatened and Critically Endangered respectively (Wallace et al., 2013b; Nel & Casale, 2015). Hawksbill turtles have been globally assessed as Critically Endangered (Mortimer & Donnelly, 2008) while green turtles as Endangered (Seminoff, 2004). Furthermore, olive ridley (Lepidochelys olivacea) turtles are rare along the coast and only enter South African waters as strays (Hughes, 1989).

Since 1963, the local conservation management agency, currently known as Ezemvelo KZN Wildlife (EKZNW), has spearheaded turtle conservation efforts in South Africa. The number of nesting leatherback turtles rose from an average of 21 per season in the first 10 years of study, fluctuating annually, to as many as 164 individual females in a single season (1994/95). Since then, the numbers have declined but stabilised at 80 and 100 females in a single season (1994/95). Since then, the number of nesting leatherback turtles has risen more consistently, from ~250 to >1,700 nests laid annually in northern KZN (Nel et al., 2013). Sea turtles have benefited from the protection of nesting beaches in the iSimangaliso Marine Protected Area since 1979 and the adjacent Ponta do Ouro Partial Marine Reserve in southern Mozambique since 2009. However, numbers of nesting turtles of both species during the last three nesting seasons have been “disappointing” (Hughes, pers. comm.).

Injured, diseased or otherwise incapacitated nesting females and turtles of other life stages are unable to function normally and may flounder at sea or wash up on shore, a phenomenon known as stranding. Strandings occur for a variety of reasons including vessel strikes, ingestion of plastic, incidental capture in fishing gear, disease, and predation by sharks amongst many others (Flint et al., 2015). Once stranded, these animals have a reduced chance of survival if not brought into human care. Understanding trends in stranded species, numbers, size class and sites, and the factors that contribute to successful rehabilitation will assist future turtle rehabilitation efforts.

METHODS

This study analysed trends in live stranded turtles admitted to the uShaka Sea World Turtle Rehabilitation Centre (TRC) in Durban, South Africa, between 2007 and 2019. Stranded turtles found along the coast of KZN are brought to the TRC by members of the public or local authorities. Records are kept of each animal, including date of stranding, species, location of stranding, and the condition of the turtle on admission. An active file for each turtle is maintained throughout their rehabilitation that details present conditions, diagnosis, and treatment as well as husbandry information. The outcome of each case is also recorded. Data on released turtles includes location, date of release, and tag information if applicable. Over time, the quality of the data has varied. Initially, data were stored in a hardcopy format. Since 2015, the data have been transferred onto the Zoological Information Management System (ZIMS, https://www.species360.org/), an international data management system designed to manage information on animals in zoos and aquaria.

Descriptive analysis of the data was undertaken using MS Excel. Animals were pooled into three weight classes: <1kg, 1-50kg and >50kg. The sex of most animals could not be determined due to immaturity and similarities in tail length. The location of the stranded animal was recorded based on one of four predetermined regions on the KZN coast from north to south: iSimangaliso Wetland Park (~186km), North Coast (~152km), Central (~80km) and South Coast (~138km) (Figure 1).
The state of the animal on arrival was classified as:

**Poor:** the animal is lethargic, little to no response to external stimulus, signs of extreme dehydration (e.g., eyes sunken)/emaciation, extensive trauma (bones visible/recent amputation/blood loss).

**Stable:** the animal reacts to external stimuli, but reactions/movements are slow, some signs of dehydration, no signs of extensive trauma/blood loss. Any amputated limbs are healing.

**Good:** the animal is hydrated (e.g., eyes not sunken in), alert, readily reacts to external stimulus, able to move easily without any encouragement.

Unless it was obvious, the cause of stranding was often difficult to ascertain. General notes on the condition of the animal on arrival were recorded including buoyancy disorder (animal could not dive), dehydration, epibiota growth on body (barnacles, leeches, etc), entanglement in fishing gear or plastic (including ingestion of plastic and/ hooks), obvious infection, external injury (body damage caused by a possible boat strike, predator attack or other), internal injury, parasites, and unknown (no visible signs of injury or illness). The three most obvious conditions for each animal were selected for this analysis. The outcome of rehabilitation was categorised as release to the wild, permanent housing in the uShaka Sea World Turtle exhibit (if the animal was unable to live independently in the wild), or death. Necropsies were performed on over 80% of turtles that had died. No turtles were euthanised.

Equipment for rehabilitation in the TRC includes large pools and tanks with filtered seawater, and a medical centre fully equipped with a digital X-ray machine, endoscope, infusion pumps, etc. Diagnoses are made through cultures, blood samples, and other procedures in the well-equipped laboratory. The TRC also receives turtles that stranded elsewhere along the South African coast. These turtles usually undergo a period of rehabilitation in other centres (such as the Two Oceans Aquarium in Cape Town) before being sent to the TRC, and hence were excluded from this analysis.

**Results**

**Biometrics**

Between 2007 and 2019, 51 turtles were admitted to the TRC: 22 green turtles, 20 loggerheads, and nine hawksbills. Eight leatherback hatchlings were recorded in two stranding events, seven of which were returned to the sea within two days and one which died; all were excluded from the analyses.
The number of turtles received by TRC each year varied between none and 10 individuals (Table 1). There was no obvious trend in the annual number of stranded turtles over the period under review, although more turtles were received in 2018 than in any other year. One adult turtle was identified as a male, while the others were too small to be positively sexed.

Nearly half (49%) of the stranded turtles weighed less than 1kg, 41% weighed 1-50kg and only three weighed more than 50kg. The weight was not recorded for two cases. The weight of stranded turtles differed among species (Table 2) where nearly all the loggerhead turtles weighed <1kg with an average weight of 46.5g. The average weight of the green turtles was 16.5kg while the average weight of hawksbill turtles was 13.1kg.

**Location of stranding**

Overall, the number of strandings per region was not proportionate to its coastline, since most stranded turtles were received from the central region, with the fewest from the iSimangaliso area. Most of the loggerhead turtles were found in the central region, while the stranded green turtles were distributed throughout the region, with the fewest from the Central region (Table 3). No hawksbill turtles were recorded from the North Coast.

**Seasonality of stranding**

There was a seasonal trend in strandings, with more turtles stranding and subsequently entering the TRC, during the summer months. This pattern is driven by small loggerhead turtles that strand between January and April. Other species strand throughout the year, with a slight peak in September for green turtles (Figure 2).

**Condition at stranding**

The turtles admitted into the TRC exhibited various conditions (Figure 3). No easily discernible cause of stranding for most of the turtles was identified. Recording of health conditions was inconsistent. For example, a turtle with an infection that resulted in a buoyancy disorder may be recorded as ‘infection’ or ‘buoyancy disorder’, while an animal admitted floating with an external injury may be recorded as ‘external injury’ since it could not be determined whether it was injured because it was floating or was floating because it was injured. Green turtles exhibited buoyancy disorders, external injuries, encounters with fishing gear and infections. The small loggerhead turtles seldom exhibited obvious conditions, although a few did have visible injuries. Hawksbill turtles were often injured on arrival. Dehydration was often noted but was usually secondary to other conditions. The growth of epibiotic on the flippers or carapace and parasites was less commonly noted and an internal injury could only be diagnosed if obvious (e.g., blood in the faeces). Plastic fragments were found in the gut during the necropsies of only one green and one loggerhead turtle, each weighing <1kg.

**Table 1. Live stranded turtles received annually by the uShaka Sea World Turtle Rehabilitation Centre, 2007-2019.**

<table>
<thead>
<tr>
<th>Year</th>
<th>Green</th>
<th>Hawksbill</th>
<th>Loggerhead</th>
<th>Total</th>
</tr>
</thead>
<tbody>
<tr>
<td>2007</td>
<td>1</td>
<td>0</td>
<td>3</td>
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<td>2008</td>
<td>1</td>
<td>1</td>
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<td>2</td>
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<td>2009</td>
<td>4</td>
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<td>1</td>
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</tr>
<tr>
<td>2016</td>
<td>2</td>
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</tr>
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<td>4</td>
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<td>2018</td>
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<td>2019</td>
<td>2</td>
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<td>0</td>
<td>2</td>
</tr>
<tr>
<td>Total</td>
<td>22</td>
<td>9</td>
<td>20</td>
<td>51</td>
</tr>
</tbody>
</table>

The number of turtles received by TRC each year varied between none and 10 individuals (Table 1). There was no obvious trend in the annual number of stranded turtles over the period under review, although more turtles were received in 2018 than in any other year. One adult turtle was identified as a male, while the others were too small to be positively sexed.

**Table 2. Weight of sea turtles rehabilitated at the uShaka Sea World Turtle Rehabilitation Centre after stranding on the KwaZulu-Natal Coast, 2007-2019.**

<table>
<thead>
<tr>
<th>Sea Turtle</th>
<th>% in Weight Class</th>
<th>North Coast</th>
<th>Central Coast</th>
<th>South Coast</th>
</tr>
</thead>
<tbody>
<tr>
<td>Green (n=22)</td>
<td>13.6 72.7 9.1 4.5</td>
<td>22.7 31.8 9.1 36.4</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Hawksbill (n=9)</td>
<td>33.3 55.6 11.1 0.0</td>
<td>33.3 0.0 44.4 22.2</td>
<td></td>
<td></td>
</tr>
<tr>
<td>Loggerhead (n=20)</td>
<td>95.0 0.0 0.0 5.0</td>
<td>0.0 20.0 65.0 15.0</td>
<td></td>
<td></td>
</tr>
</tbody>
</table>

**Table 3. Location of rehabilitated at the uShaka Sea World Turtle Rehabilitation Centre after stranding on the KwaZulu-Natal Coast, 2007-2019.**

<table>
<thead>
<tr>
<th>Sea Turtle</th>
<th>% at Each Location</th>
</tr>
</thead>
<tbody>
<tr>
<td>Green (n=22)</td>
<td>22.7 31.8 9.1 36.4</td>
</tr>
<tr>
<td>Hawksbill (n=9)</td>
<td>33.3 0.0 44.4 22.2</td>
</tr>
<tr>
<td>Loggerhead (n=20)</td>
<td>0.0 20.0 65.0 15.0</td>
</tr>
</tbody>
</table>
**Outcome of rehabilitation efforts**

Of the turtles that entered the TRC, over half were released, while 14.0% could not be released due to the severity of their injuries. These turtles were placed on exhibit in the uShaka Sea World aquarium. Death was a more frequent outcome for turtles that arrived in poor condition, while 91.0% and 83.3% of those that arrived in a stable or good state respectively were released. A total of 18 turtles died (35%) during rehabilitation. There was little difference in the response rate of rehabilitation among species (Table 4).

The outcome of the rehabilitation process did not appear to be related to the weight of the turtle on arrival (Figure 4).

For all the species, the average time spent in rehabilitation was 208 days (range: 1-1,485 days). Ten turtles died (35%) during rehabilitation.

**Table 4. State of arrival and rehabilitation outcome for sea turtles rehabilitated at the uShaka Sea World Turtle Rehabilitation Centre after stranding on the KwaZulu-Natal Coast, 2007-2019.**

<table>
<thead>
<tr>
<th>State on Arrival</th>
<th>Rehabilitation Outcome (%)</th>
</tr>
</thead>
<tbody>
<tr>
<td></td>
<td>Died</td>
</tr>
<tr>
<td>Poor</td>
<td>50.0</td>
</tr>
<tr>
<td>Stable</td>
<td>0.0</td>
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<tr>
<td>Good</td>
<td>0.0</td>
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</tbody>
</table>

**Species**

- **Green**: 40.9 | 9.1 | 50.0
- **Hawksbill**: 22.2 | 11.1 | 66.7
- **Loggerhead**: 35.0 | 20.0 | 45.0

For all the species, the average time spent in rehabilitation was 208 days (range: 1-1,485 days). Ten turtles died (35%) during rehabilitation.
turtles spent <10 days in the TRC. Green turtles spent an average of 199 days in the TRC. Excluding one turtle that spent 4 years undergoing rehabilitation, loggerhead turtles spent an average of 190 days in the TRC, while hawksbill turtles spent 125 days in the TRC.

**DISCUSSION**

The three species of turtles (loggerhead, green and hawksbill) that were brought into the TRC during the study period varied in stranding location, weight, condition, duration of rehabilitation and eventual outcome. The small sample size and high variability made it difficult to interpret the success of treatments administered. However, pertinent observations are discussed as they may contribute to an improvement in our understanding of turtle stranding and rehabilitation.

There was no clear trend in the number of turtles in total or per species admitted per year. Although Flint *et al.* (2015) noted an increase in the number of turtles being sent to rehabilitation centres in Australia over time, this was not apparent at TRC despite improved public awareness of what to do when encountering a stranded turtle. uShaka Sea World has publicised the plight of turtles on-site in aquarium signage and through social and print media. The formation of a 'Stranding Network,' coordinated by the local conservation authority (Ezemvelo KZN Wildlife) and made up of individuals who have been trained in how to handle stranded animals has also contributed to better handling of stranded animals. Numerous organisations (including Ezemvelo KZN Wildlife, lifeguards, the National Sea Rescue Institute, the South African Police Services, conservancies, etc.) have staff working on or near popular beaches. Staff have attended workshops facilitated by uShaka Sea World and are now trained as first responders and follow national protocols for stranded animals. It was thought that this increase in awareness and capability of response would have resulted in an increase in reported strandings of live turtles. However, the numbers of live stranded turtles of each species may be more closely linked to the population size of turtles at sea rather than human interventions. As an indication of the prevalence of different turtle species along the KZN coast, the bycatch of the bather protection nets (shark nets) were analysed. Interestingly, loggerhead turtles were the most frequently caught (67%) in the shark nets along the KZN coast between 1981 and 2008, where green (19.6%), leatherback (8.8%) and hawksbill (3.1%) turtles made up the rest of the bycatch. Most of the animals caught were classified as immature, however they were all large enough to be caught in the large meshed gill nets (25 cm bar) (Brazier *et al.*, 2012). Given their prevalence in the area it is surprising that larger loggerhead turtles were not brought into the TRC.

The location of live stranded turtles is more likely to be a function of human presence along the coast rather than the absolute number of animals stranding. Most turtles were received from the Central region, an area with a high level of coastal development and a high associated human population density. Although both loggerhead and leatherback turtles nest on the beaches of the iSimangaliso Wetland Park, neither of these species were reported stranding from those beaches. Except for public access points such as Sodwana Bay, Cape Vidal and St Lucia, the iSimangaliso Wetland Park beaches are remote with few people, so turtle strandings are often likely to go unreported. In this study, small loggerhead turtles were primarily found stranded in the central part of the coast. It is likely
that these small turtles were post-hatchlings that had drifted southwards in the Agulhas Current after hatching on the beaches of the iSimangaliso Wetland Park and Ponto do Ouro Partial Marine Reserve. This would also account for the seasonal nature of their stranding between January and April. This pattern of stranding has been noted since the early 1970’s (Hughes, 1974).

Not surprisingly, the turtles from the central KZN coast were slightly smaller than those found stranded further south (Ryan et al., 2016), as it would have taken slightly longer for them to reach the more southern beaches. The green and hawksbill turtles that were rescued along the whole KZN coast ranged in size. Both species stranded throughout the year.

The duration of rehabilitation in the TRC was high in comparison to a study in Australia where 35% of turtles were released within 28 days of arrival and average days in care decreased from 392 in 1999 to 84 in 2013 (Flint et al., 2017). However, it was similar to the findings from Florida, USA, where time in rehabilitation varied between one year and more than three years (Baker et al., 2015). In our study, rehabilitation success did not increase with body size as was found by Baker et al. (2015).

As has been noted in other regions (Flint et al., 2017), the primary cause of stranding is usually difficult to ascertain. Many turtles exhibited multiple conditions, e.g. buoyancy disorders, infection, and external wounds. Buoyancy disorders, often caused by gas trapped within the intra-coelomic cavity, render the turtles unable to dive and feed. Such animals eventually strand in a weakened state (Mettee, 2014). Damaged lungs, infections, intestinal blockages, and stress can also cause turtles to become buoyant. Our study found that buoyancy disorders were prevalent in green turtles. Dehydration was common, although this was often overshadowed by more serious traumas. The range of conditions on arrival recorded at the TRC was similar to that noted by other studies (e.g., Flint et al., 2017) where disease, buoyancy disorder and fracture were most commonly noted.

It was generally easier to attribute the cause of stranding for turtles that showed obvious signs of an encounter with fishing gear or had external wounds. Turtles can become entangled in fishing gear, caught as bycatch, swallow fishhooks or line, or become damaged after impact injuries caused by vessels. The inshore habitats of the loggerhead and green turtles make them particularly vulnerable to fishing gear impacts, as most fishing effort is expended closer to the coast (Everett, 2014). The incidence of direct interaction with fishing gear was far lower in our study than that noted by previous studies (Poli et al., 2014; Nelms et al., 2016). This difference is likely due to the relative lack of large commercial fishing operations off the KZN coast compared to other areas (Everett, 2014).

Previous studies have noted ingestion of plastic pollution to be a threat to turtle survival (Hoarau et al., 2014; Nelms et al., 2016). However, unlike a study conducted in the southern Cape, South Africa, where 60% of loggerhead post-hatchlings that died within two months of stranding had ingested plastic fragments (Ryan et al., 2016), plastic pollution was only noted in two turtles in this study.

Half of the stranded turtles in this study were released. Those that could not be released, as they would not be able to survive independently in the wild, due to injury or poor condition, were placed into the uShaka Sea World Turtle Exhibit. These success rates compare favourably to those in Queensland, Australia where 35% of the study animals were released (Flint et al., 2017) and in Florida, USA where 36.8% were released (Baker et al., 2015).

Treatments varied with the condition of each animal upon arrival and the individual animal’s response to treatment was different. Each action, response, and outcome were monitored and recorded to ensure that findings could be used in future cases. The state of the animal on arrival was one of the primary predictors of the success of the rehabilitation. Since over half of the animals that arrived in a poor state died, the efforts expended on these individuals should be weighed against their chance of survival. Where resources (both time and financial) are limited, it may be wise to critically assess the chance of survival for each individual prior to commencing a lengthy process of rehabilitation. However, the large variability in most factors related to the outcome makes it a difficult decision. In our study, some of the most compromised animals survived, while some of those that appeared to be in a relatively good condition died. Perhaps the efforts expended on each animal are justified, should the resources (time and financial) be available.

Each animal undergoes a thorough health assessment prior to release from the TRC. However, turtles that require extended periods in rehabilitation may not be the best candidates for release. During the rehabilitation period they may have been exposed to pathogens which they may transmit to the wild population on release (Baker et al., 2015). More research is warranted to determine the survival of turtles that are released.

Although release to the wild is the goal for every rehabilitation effort, the turtles that cannot be released can play a vital role in education and conservation (Peck & Hamann, 2013; Baker et al., 2015; Martin et al., 2015). Research undertaken at the uShaka Sea World showed that the emotional connection generated between an animal in human care and visitors can be a contributing factor to future pro-environmental behaviour (Mann et al., 2018). After a turtle encounter, visitors expressed an
intention to undertake pro-environmental behaviours such as reducing, re-using and recycling (Mann & Ngcobo, 2017). When guests were asked what they would remember most from their interaction, they reported that the experience of being close to the animal was the most memorable. It has also been suggested that the high levels of excitement brought about by interacting with animals allows for higher levels of learning and retention (Mann et al., 2020). In addition, social media posts about rehabilitation of stranded turtles generated greater engagement by the public than other animal rehabilitation posts (Mann & Zwane, 2019). This highlights the important role rehabilitation plays in sensitising the public to the plight of turtles and encouraging appropriate environmental behaviour.

Limitations

Analysis of a data series collected over time can be challenging if the data has not been recorded consistently and systematically. In this study there were a few inconsistencies with respect to data collection as the data was collected by different people over a period of time. The inconsistencies were compounded by missing data. A new system is now in place with standardized data collection fields that should help facilitate future analyses.

CONCLUSION

There is considerable debate on whether resources and skills would be better directed towards implementing strategies to mitigate the causes of turtle stranding, rather than treating injured animals, often with low success rates (Baker et al., 2015; Flint et al., 2017). Indeed, much work is required to reduce the negative impact of humans on turtles. However, we believe that rehabilitation of injured sea turtles can have positive outcomes, including improved understanding of rehabilitation practices and greater success, increased public support for marine conservation and pro-environmental behaviour, and increased numbers in wild populations after releases. We recommend further research on all these aspects.

ACKNOWLEDGEMENTS

The TRC is operated by SAAMBR, a non-government, non-profit company responsible for the operation of uShaka Sea World, the largest aquarium in Africa, situated in Durban, on the east coast of South Africa. SAAMBR has contributed to turtle rehabilitation since the 1990s. Special thanks to the SAAMBR Quarantine, Animal Health and Curatorial teams for their commitment to animal care and welfare. Dr Francois Lampen is thanked for his review of this manuscript. Tracy Shaw, Lyn Britz, Sheila Lang and Bruce Mann are thanked for their support and assistance with the paper. An anonymous reviewer is thanked for their valuable comments.

Literature cited:


**INTRODUCTION**

Initially developed as an instrument for wildlife photography, camera traps were subsequently used in hunting and have now transformed into a conservation tool (Kucera & Barrett, 2010). Camera traps allow us to observe activities taking place in the wild with minimal intrusion and have many current and potential applications in sea turtle research and conservation.

Any camera that is not triggered by a human (instantly or at a pre-set time) is a camera trap, although some studies that use the term include pre-programmed cameras. In the past, camera trap studies have primarily focused on terrestrial mammals, exploring behavioural patterns as well as their presence in certain habitats. Such methods allowed researchers to collect relatively unbiased data for long periods of time. With technological advances, increased availability, and reduced prices, the popularity of camera traps as a research tool grew and they were adopted to study a variety of species.

For camera trap studies to be viable, it is necessary for researchers to know the exact area in which the target animal is expected, to ensure that it will trigger the camera trap. As terrestrial phases of the sea turtle life cycle are confined to predictable regions of nesting beaches and areas immediately adjacent to known nest locations, camera trapping is a viable method to study turtle biology and threats during nesting, egg incubation and hatching.
emergence; camera traps could also be used during in-water observational and monitoring projects. This paper demonstrates the current and potential application of camera traps in sea turtle research and conservation using examples from studies of freshwater and marine species of turtles, then reviews the technical aspects of deploying camera traps in terrestrial and marine environments.

APPLICATIONS OF CAMERA TRAPS ON NESTING BEACHES

Monitoring nesting sea turtles

There are no reports of triggered camera traps being deployed to monitor nesting beaches at this time. However, time-lapse beach photography projects using pre-programmed cameras with trap capabilities have been employed using two approaches. One is to position cameras at sites where much or all of the beach is visible, for example on a headland or sand cliff overlooking a cove, and to program the camera to take photos every morning and record fresh new tracks made by nesting turtles (S. Whiting, pers.comm.). Another approach is to use the camera to take pictures at intervals to record how many days turtle tracks remain visible on the beach. When used for such a purpose, however, care is needed to ensure that representative beach microhabitats are monitored to account for the impact of different environmental conditions on track longevity. At select beaches at Diego Garcia and Nelson Islands in the British Indian Ocean Territory, time lapse photography is being used to inform beach monitoring frequency and improve estimates of nesting activities in the study locations (Esteban & Mortimer, 2018; Wood et al., 2019). Similar approaches could be used at other sites, depending on physical characteristics of the nesting beach and surrounds.

Identifying predators of nesting sea turtles, eggs and hatchlings

Camera traps can be used to complement findings from other methods of identifying predators, such as trackboards (Buzulecü et al., 2016), scat analysis (Dawson et al., 2016) and physical observation (Doody et al., 2009; Erb & Wynken, 2019; Unger & Santana, 2019). However, the potential for cameras to introduce bias and affect rates of predation by attracting or deterring some predator species from their normal behaviour (e.g., Richardson et al., 2009) should also be taken into consideration.

Animal predation on nesting turtles is rare, and throughout countries in the Indian Ocean and Southeast Asia it may be limited to isolated incidents involving saltwater crocodiles (Whiting & Whiting, 2011) and hyenas (Olendo et al., 2016). In the event that predation on nesting sea turtles by these or other species does increase, camera traps could give insight into predator behaviour. For example, Guilder et al. (2015) and Escobar-Lasso et al. (2016) determined the importance of sea turtles as a dietary item to jaguars (Panthera onca) in Costa Rica, as well as jaguar feeding and scavenging behaviour, using camera traps.

While predation on nesting sea turtles might be rare, depredation of nests is an ongoing concern in the same region (Ekanayake et al., 2002, 2010; Islam et al., 2002a,b; Shanker & Choudhury, 2006; Ficetola, 2008; Tripathy & Raiasekhar, 2009; Thi et al., 2011; Whiting & Whiting, 2011; Salleh et al., 2012; Ellepola et al., 2014; Mancini et al., 2015; Nasher & Al Jumaily, 2015; Olendo et al., 2016; Phillott et al., 2018a; Williams et al., 2019). To mitigate this threat, eggs are often relocated to a fenced area commonly known as a hatchery (Salleh et al., 2012; Abd. Mutalib & Fadzly, 2015; Phillott, 2018; Phillott & Kale, 2018, Phillott et al., 2018a,b; Howard et al., 2019). However, best practices in the collection, transport and incubation of eggs and handling of hatchlings have to be followed to reduce risks to embryo survival and hatching fitness (reviewed by Phillott & Shanker, 2018), and require economic and human resources that might not be available to local conservationists. An alternative to reducing predation of eggs and hatchlings by relocating them to a hatchery is protecting nests in situ.

As the appropriate method for protecting sea turtle eggs in their original location on the nesting beach can depend on the species of predator (reviewed by Phillott (2020) in this issue of IOTN), tools to identify animals depredating nests are also required. Predators can potentially be identified from their tracks and patterns of digging into a nest (e.g., Gandu et al., 2013; Korein et al., 2019) but these signs might actually be created by scavenging behaviour or secondary predation after earlier predators have opened the nest (e.g., Barton & Roth, 2008).

Camera traps have proved effective in helping researchers identify species that pose a threat to sea turtle eggs. The use of camera traps to monitor artificial nests was first popular amongst ornithological studies and has been adopted to understand sea turtle predators. Maier et al. (2002) studied the depredation of artificial freshwater turtle nests using subterranean triggers to activate the shutter of a 35mm film camera. The triggers were installed within the nest chamber, connected by a trigger wire to a camera facing the entrance to the nest. It effectively captured images of predators such as racoons (Procyon lotor), striped skunks (Mephitis mephitis), gray foxes (Urocyon cinereoargenteus), and fiders (Martes pennanti). Motion-triggered camera traps have also been used in such studies; for example, artificial nests of alligator snapping (Macrochelys temminckii) turtles near the primary nesting area were monitored to identify and quantify the
The relative contribution to nest depredation by raccoons (*Procyon lotor*), armadillos (*Dasypus novemcinctus*), opossums (*Didelphis virginiana*), bobcats (*Lynx rufus*) and otters (*Lontra canadensis*) (Holcomb & Carr, 2013).

Camera traps have also been used to identify predators of hatchlings. Erb & Wynenek (2019) investigated the nest-to-surf mortality of loggerhead (*Caretta caretta*) sea turtle hatchlings by combining techniques of camera trapping, direct observation and hatchling track maps. The camera traps were placed behind nests and programmed to take an image every 5 to 10 seconds using the time lapse mode, recording predation events by ghost crabs (*Ocypode quadrata*), night herons (*Nyctanassa violacea*) and gray foxes. Bieber-Ham (2010) used camera traps to identify raccoons and opossums as predators and monitor their feeding on painted, plaster-cast turtles (*Chrysemys picta*) hatchling replicas. Giuliano *et al.* (2014) used camera traps to film and photograph nocturnal depredation on flatback (*Natator depressus*) sea turtle hatchlings by nankeen night herons (*Nycticorax caledonicus*) and black-necked storks (*Ephippiorhynchus asiaticus*), the first steps in assessing the impact of avifauna predation on turtle population dynamics.

**Determining the behavioural patterns of predators**

The behavioural patterns of predators, including foraging times and cues used to find nests, can also be studied using camera traps. Anyone planning this type of study might find the review of camera trapping for conservation behaviour research by Caravaggi *et al.* (2017) helpful to read.

The characteristics of loggerhead sea turtle nest visitations by lace monitors (*Varanus varius*) and yellow-spotted monitors (*V. panoptes*) were studied using camera traps (Lei & Booth, 2017a,b; Madden Hof *et al.*, 2020). By capturing motion-triggered still images and metadata (time and date), the number and frequency of visits in different time frames within a day (Lei & Booth, 2017a) and the potential disturbance caused by a human presence; the use of a camera trap also allows uninterrupted data collection while being less labour intensive.

Camera traps were also used to test the efficacy of different raccoon excluder devices on simulated diamondback terrapin nests (Buzuleciu *et al.*, 2015), allowing researchers to understand why some cage features were more successful than others without the potential interference that may be associated with direct observations.

**Surveillance for illegal take of eggs**

Recently, camera traps have also emerged as a covert and relatively inexpensive surveillance tool, monitoring remote regions to detect the illegal take of sea turtle eggs without the need for regular patrols by rangers (Wearn & Glover-Kapfer, 2017). Camera traps on nesting beaches can identify those involved in the illegal take of eggs and collect evidence against them. The use of networked camera traps would also allow preventive measures to be taken when illegal take is detected. As camera traps used for this purpose are at a high risk of theft, equipment must be as covert as possible as well as located at a height that captures identifiable features of responsible persons (Wearn & Glover-Kapfer, 2017).
In-water studies of sea turtles

Due to the failure of sensors underwater, remote exploration of the aquatic realm using camera traps has been limited (Wearn & Glover-Kapfer, 2017). Instead, studies have used submerged underwater cameras to record continuous videos and hence gain an insight into the activities of marine organisms. The use of Baited Remote Underwater Video (BRUV) systems has allowed researchers to study marine species diversity (Osgood et al., 2019) and behaviour (Bond et al., 2012) by recording the organisms which were attracted to the bait. Favaro et al. (2012) developed a modified version of this, known as the TrapCam, which was also effective in obtaining in situ observations of marine animals at depths up to 100m and could be modified to understand sea turtle interactions with deep-water fishing gear.

Recent innovations in camera trap systems have, however, proved promising in capturing remotely triggered images of underwater phenomena. An underwater stereo camera such as the TrigCam can be programmed with an algorithm to record images whenever a predefined change in pixels is detected. The technology allows researchers to tailor their study to target wildlife of a specific size (Williams et al., 2014), and may be useful in studies of the in-water behaviour of animals such as sea turtles.

TECHNICAL ASPECTS OF CAMERA TRAPS

Camera and trigger features

When choosing a camera trap, care must be taken to ensure the features of the trap model are compatible with the specific nature of the study. Newey et al. (2015) provides a user’s perspective on the deployment, operation and data management when using more affordable ‘recreational’ models in comparison to expensive ‘professional’ models which could help novices in camera trap usage in their decision about which model to purchase. Features of the camera and trap trigger as described below should also be considered.

As the quality of data captured is dependent on the effectiveness of the trigger system, the trade-off between availability, affordability, and suitability of camera traps with desired features must be considered. The target animal for the study will also help determine whether a camera trap should employ an indirect or a direct trigger system.

An indirect trigger system- which senses the presence of the animal in the vicinity of the camera trap via movement or heat signatures- is ideal when the target animal is large and endothermic, like feral pigs or dogs. Of these, a passive infrared (PIR) trigger is the most suitable considering the large body mass and heat signal of many predators, and most commercially available camera traps have PIR triggers as the market is driven primarily by its demand for deer scouting and hunting. These trigger systems are also more concealable and less startling (Wearn & Glover-Kapfer, 2017). An emerging technology for underwater camera traps also employs an indirect trigger system. It employs a software algorithm to trigger recording when an animal appears by detecting a change in pixels. The algorithm may be tailored to capture animals of a particular size and, hence, reduce the possibility of unwanted shots (Williams et al., 2014).

Smaller mammals and reptiles may not have a heat signal strong enough to trigger a PIR (Eskew 2012; Hobbs & Breitme, 2017) and might require the use of direct triggers. These include mechanical triggers which can be installed within the nest chamber such that the camera trap is triggered only when directly pushed or pulled by the animal during depredation or when the nest is otherwise disturbed. Options include tilt switches (as in Maier et al., 2002), trip wires, pull wires, pressure plates, and active IR (AIR) triggers (Tuberville & Burke, 1994). The most modern direct trigger, AIR sensors require the animal to move through a predictable path such that it disturbs an IR beam between a transmitter and receiver. However, in addition to being less commercially available, these are also more visible and intrusive and many studies have found PIR camera traps to be effective, even in the cases of reptiles like monitor lizards (Beukeboom, 2015; Lei & Booth, 2017a,b; Madden Hof et al., 2020) and small mammals like rats (Gronwald et al., 2019). Many camera traps have the option of increasing PIR sensitivity, which would increase the likelihood of capturing ectotherms (Wearn & Glover-Kapfer, 2017).

In addition to motion-triggered image capture, the time lapse feature of camera traps can also be used to record potential predators in the nest environment at regular intervals (Geller, 2012; Beukeboom, 2015; Erb & Wyneken, 2019). This setting would ensure that images will be taken even if the trigger does not detect the presence of a predator; however, it produces a huge volume of images for analysis.

It is also important to consider the detection zone and field of view of the camera. These are integral to the study as the detection zone is the range within which movement must occur to trigger the camera and the field of view of the camera is the area that will fall within the frame of the image. If the study is investigating nest predation, a detection zone that is narrower than the field of view would be appropriate as it would prevent accidental triggers and empty shots, ensuring all the shots taken include the target i.e. the predator at the nest (Trolliet et al., 2014). However, if the study intends to include nesting behaviour or hatchling predation, a wide
Detection zone would be important as the target activity may occur beyond the immediate nest. The trigger speed of the camera and the relay time (delay between each shot) should also be factored in when choosing a camera trap. A fast trigger speed can compensate for a narrow detection zone (Rovero et al., 2013).

The camera must also have a battery capacity sufficient for it to be deployed for potentially extended periods of time. This is dependent on the number and type of batteries used as well as the energy efficiency of the camera. Based on the battery life as well as the capacity of the memory card, the camera trap may have to be regularly visited and items replaced. In cases of remote camera trapping locations, networked camera traps may be deployed which can send almost real-time data to the researcher. Though these are expensive, they also allow the data to be regularly backed up such that in case of any damage or theft, the data is not lost (Wearn & Glover-Kapfer, 2017).

Cameras deployed at night on nesting beaches should have the feature of IR flash. While white flashes can produce coloured images as opposed to the monochrome images from IR flashes, the former may deter predators and introduce bias while IR flashes are invisible to most animals. A visible white flash could also attract human attention to the camera trap and increase the risk of theft (Wearn & Glover-Kapfer, 2017), or temporarily misorient hatchlings. In specific cases where the red glow of IR frightens predators, such as coyotes (Eskew, 2012), no-glow (black) IR cameras can be deployed which emit a wavelength that is almost undetectable (Wearn & Glover-Kapfer, 2017).

If images are only to be used for detection and identification of predators, lower quality images would suffice but if the collected images may be used for campaigns or conservation awareness programs then higher resolution images will be required. In cases where individual predators need to be identified, a higher resolution may be necessary to discern unique features. Additionally, if the study requires insights into behavioral aspects, the video feature available in many camera traps would be helpful (Giuliano et al., 2014; Gronwald et al., 2019).

**Camera mount and position**

Camera traps in a beach environment may be mounted on a tripod, wooden stakes, metal t-posts (Urbanek & Sutton, 2019) or PVC pipes (Eskew, 2012). The mount structure must be sturdy enough to carry the weight of the camera trap and ensure it does not move. If there is a nearby tree or pole, these could be ideal mounts. One of the biggest challenges when using camera traps is reducing the possibility of theft. To prevent theft, the camera traps can be placed within commercially available housing cases that can only be opened with a special key. These housings also protect the cameras from damage by animals. Most anti-theft cases are intended to be attached to trees or poles that cannot be removed. However, these are often difficult to find in the sea turtle nesting environment. Deploying the camera traps only at night may reduce the risk of theft (Shipman, 2019); however, this schedule provides limited data and requires regular installment and removal.

Camera traps are designed to be robust and weatherproof, however most of them are not built for a beach environment. It may be necessary to regularly clean the camera traps of sand. To prevent damage due to humidity, desiccation packs can be placed in the camera trap or within its casing (King, 2016). The casing should have seals that prevent entry of rain, dust, sand and insects etc. Depending on the target predator, the camera trap must be installed at an appropriate height to ensure that the animal will fall within the detection range as well as the field of view of the camera. They can be tilted slightly downwards to avoid triggers due to sunrise and sunset. Additionally, unless required, the camera trap detection range should not include the ocean as this may lead to waves and tidal movements acting as triggers (King, 2016). This is important to make sure the memory card is not exhausted due to repetitive empty shots.

**Data analysis**

Camera trapping surveys often produce large amounts of data (still images or video) that are cumbersome to analyse. Using a PIR that senses movement can result in a large proportion of images that are empty shots due to detection of non-animal movement (false triggers) such as the movement of foliage. Many studies sort the images manually; however, emerging camera trap data analysis software that weigh pixel variations against the background to filter out images void of animals can be used to streamline the process of analysis (see Hobbs & Brehme, 2017; Wearn & Glover-Kapfer, 2017).

**ETHICAL CONSIDERATIONS WHEN USING CAMERA TRAPS**

In environments where there is a high degree of human activity, projects must consider the privacy of local residents. Steps should be taken to avoid non-consensual monitoring and surveillance. For example, local communities should be informed about the purpose, general location, and operation of cameras before traps are installed. Projects can also implement community-based conservation, compensating locals for regularly checking the camera traps, retrieving memory cards, and replacing batteries. Additionally, there must be a plan to respectfully delete any accidental images of people.
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PROTECTION OF IN SITU SEA TURTLE NESTS FROM DEPREDATION

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INTRODUCTION

Nesting turtles act as biological transporters of nutrients from marine to terrestrial ecosystems, where eggs and hatchlings eaten by terrestrial predators contribute to coastal food chains and nutrient cycles (see Bouchard & Bjorndal, 2000; Madden et al., 2008). Depredation of sea turtle nests should, therefore, only be of concern if the affected population is categorised as Threatened, if it is occurring at or has the potential to reach unsustainable levels, if the predator is introduced or reintroduced to the area, or if the increasing population of one predator is the result of control measures against another. Predator management may also be desired if local tourism relies on the presence of sea turtle nests.

A common strategy for protecting eggs and/or hatchlings from excessive predation is to immediately relocate clutches to a hatchery, a protected area enclosed by a fence to reduce animal entry. However, care must be taken to ensure that collecting, handling, and incubating eggs does not itself reduce the number of eggs that hatch (Phillott & Shanker, 2018). A viable alternative to moving eggs to a hatchery is protecting them in their original position. In situ protection reduces the potential risks associated with collecting and moving eggs if best practices cannot be implemented due to restricted resources. Nests would still require regular inspection by project personnel or local community members to ensure continued protection. While sea turtle festivals and other environmental education initiatives could continue, participants may need to walk further to reach the nest location.

At nesting beaches throughout the Indian Ocean and Southeast Asia (IO & SEA) region, hatchery programs are commonly and successfully employed to control predation rates. However, in situ nest protection and predator management strategies can also be highly effective, as proven by reduced nest depredation rates on the east coast of Florida, USA, from 95% to <10% (Engeman et al., 2005). Therefore, this paper reviews methods for in situ protection of sea turtle eggs and hatchlings from predators found in IO & SEA countries. While the discussed predators have been limited to those in the region, the strategies for protecting nests have been drawn from studies around the world so conservationists can consider the various potential options and determine what best suits their needs. Before implementing in situ nest protection, the animal/s predating on eggs and hatchlings should first be identified by analysing scats (Brown & Macdonald, 1995; Blamires, 2004), stomach contents (Hilmer et al., 2010; Engeman et al., 2019), tracks in the substrate, digging patterns used to expose nests (e.g., Drake, 1993; Tripathy & Rajasekhar, 2009; Gandu et al., 2013), visual observations (Tripathy & Rajasekhar, 2009), and/or camera trapping (reviewed in this issue by Kotera & Phillott, 2020). It is important to distinguish between animals that predate on eggs and hatchlings in undisturbed nests and those that scavenge eggs which have been exposed by other species in order to choose the most appropriate management strategy.

STRATEGIES FOR REDUCING DEPREDATION

Actions to manage depredation can focus on predator
control (see below), and/or removal of debris and other obstacles to reduce the time taken by vulnerable hatchlings to cross the beach (Burger & Gochfeld, 2014). It is also not advised that hatchlings emerging from nests protected in situ or in hatcheries be held or released in batches of more than one clutch at a time. If this must occur under exceptional circumstances, then personnel should release hatchlings at different times of day and locations to reduce the likelihood of creating ‘feeding stations’ for predators (Mortimer, 1999).

To manage heavy depredation rates that are reducing population resilience, projects may need to conduct an economic analysis to determine which approaches will result in the greatest benefit in terms of egg and hatching survival, ongoing ecotourism, ecological and cultural benefits etc., for the budget available (e.g., Engeman et al., 2002, 2010). Consulting experts in management of the predator species can dramatically improve the success of intensive management projects (Engeman et al., 2011).

Removing predators from nesting beaches and adjacent areas

Baiting, trapping and removal, and/or neutering (desexing) can be used to reduce predator population numbers so predators are effectively removed from the beach and adjacent areas (e.g., Algar et al., 2011a,b; Dias et al., 2017; Leo et al., 2018). However, removal efforts need to be ongoing; when predator removal at Hobe Sound National Wildlife Refuge FL, USA, was stopped part-way through a nesting season, depredation rates increased to 1.5-3.0 times than before management was initiated (Engeman et al., 2006). Planning a predator removal project should also consider that removal of a predator species could result in another species taking its place. For example, the use of 1080 (sodium fluoracetate) poison baits was successful at reducing fox (Vulpes vulpes) depredation of nests at Wreck Rock QL, Australia, but allowed monitor lizards (Varanus spp.) to utilise the vacated niche and become the dominant predator (Lei & Booth, 2017). In another case, the removal of racoons (Procyon lotor) from beaches in Florida, USA, resulted in increased depredation of hatchlings by ghost crabs (Ocypode spp.), whose individual size and population number were previously kept in check by the racoons (Barton & Roth, 2008).

Excluding predators from nesting beaches

Fencing to exclude predators from the nesting beach has recently been suggested for Gahirmatha, India (e.g., Behera & Kaiser, 2020). However, fences have the potential to entangle nesting sea turtles and reduce the access of terrestrial species and local communities to beach resources, and hence should only be considered after local consultation and with appropriate permits or approval from relevant authorities.

Excluding or deterring predators from nests

Depredation is most likely to occur soon after oviposition, in the period just before hatching, and at hatchling emergence. Clutches laid closer to vegetation and in areas of greater nest density may be at greater risk (see Leighton et al., 2011). Therefore, projects with limited resources may choose to focus on protecting the most vulnerable nests or nests during the most vulnerable time.

Chemical deterrents

Various chemicals have been trialled to deter predators from digging into sea turtle nests. Wolf urine (Canis lupus) dispensers on nesting beaches deterred coyotes (Canis latrans) from depredating nests (Wauson, 2019). Also effective against coyotes was the powder of red savina habanero peppers (Capsicum chinense; 400,000–500,000 Scoville units) when applied to the sand surface in a 0.5m radius circle around the nest and covered with a thin layer of sand to minimise disturbance by wind or rain. However, the same powder did not reduce depredation rates when applied to sand ~3cm above the top egg in the nest (Lamarre-DeJesus & Griffin, 2013). Similarly, chilli powder (heat level 10 of 10) applied to a 0.5 x 0.5m square around loggerhead (Caretta caretta) nests at a depth of 10cm below the beach surface was not a deterrent to monitor lizards (Varanus spp.; Lei & Booth, 2017). Habanero and other pepper or chilli powders may be a more effective deterrent for predators which rely on olfactory cues to detect nests (e.g., coyotes) than species that use a combination of olfactory, visual and/or tactile cues (e.g., racoons) (Lamarre-DeJesus & Griffin, 2015). Projects that want to use pepper/chilli powder as a predator deterrent should remember that effects of the irritant on hatchlings and the predator species have not been studied.

Visual deterrents

Flags have had mixed success in deterring predators from nests. White flags (50 x 80cm), printed with the project name in red and staked so that the flag blew across the top of the nest, successfully decreased predation of loggerhead nests in northeast Brazil (Longo et al., 2009; see Figure 2 in the paper). However, red flags (30 x 40cm) on a stake ~70cm above beach surface and ~30cm from nest were not effective against monitor lizards (Lei & Booth, 2017). A study in the use of flags to mark the position of freshwater turtle nests did not find that racoons began to associate flags with a food source (Tuberville & Burke, 1994; Burke et al., 2005), but this potential should be investigated with different predators before flags are extensively used to protect nests. The effectiveness of flags in deterring predators...
from nests would also rely on suitable wind conditions.

Audible deterrents

Only one study has investigated the use of audible deterrents on potential predators of sea turtle eggs. A metal rattle sewn into the base of white flags described by Longo et al. (2009; see Figure 2 in the paper) did not improve their deterrence efficiency against foxes.

Nest enclosures

Nest enclosures may be above ground or buried below the beach surface, vertically surrounding and/or horizontally covering the clutch of eggs and surrounding area. Types of enclosures include grids, panels, baskets, and cages, and designs suitable for different types of predators are described below. While the type and design of a nest enclosure may differ depending on type of predator and available materials, several features of the material used to make all enclosures are important:

1. The mesh size should allow hatchlings to escape the enclosure unaided and without risk of entanglement, ~50mm for loggerhead, green (Chelonia mydas), hawksbill (Eretmochelys imbricata), and olive ridley (Lepidochelys olivacea) turtle hatchlings, and ~70mm for leatherback (Dermochelys coriacea) and flatback (Natator depressus) turtle hatchlings. Sometimes, a smaller mesh size is required for the predator (e.g., ghost crabs) or is the only material available. In this case, nest covers buried or at the beach surface level can include a detachable disc that is removed a week before the predicted emergence date so hatchlings do not become trapped (e.g., see Yerli et al., 1997). If the nest enclosure is elevated above the beach surface, then it should be removed at dawn and replaced at dusk when most predator activity increases. This will reduce the risk of hatchlings overheating if emergence occurs during the daytime.

2. The material should not disrupt the hatchlings magnetic imprinting. Magnetically inert metals (such as aluminium), plastic, or wood are suitable (Irwin et al., 2004).

CONTROL OF SPECIFIC PREDATORS

The same nest enclosure may not be effective against different predators. For example, cages that effectively reduced racoon predation at Keewaydin Island in the USA were not effective against wild/feral pigs (Sus scrofa) (Engeman et al., 2016). Hence, effective strategies have been summarised by predator species or taxa below.

Ants

Solenopsis spp. (West, 2010) and an unknown species (Kelaskar et al., 2016; Arun, 2019) of ants have been reported as invading sea turtle nests, predating on late-stage embryos in pipped eggs, and/or predating on or stinging hatchlings in the IO & SEA. Ants may be attracted to the scent of disturbed sand or secreted mucous during the nesting process or environmental conditions within the egg chamber (Allen et al., 2001). Descriptions and images of egg invasion of nests and penetration of eggs are available in Ikaran et al. (2020).

The effects of ant depredation on eggs is likely to be low at a population level (Holbrook et al., 2019). Ants enter nests via underground foraging trails (Buhlmann & Coffman, 2001), so control of recurring or serious infestations can be challenging because they may not be visible on the beach surface (Kelaskar et al., 2016). Infested nests in Malaysia have been excavated and unaffected eggs relocated to a different position (Chan, 2013). Ant control measures reported as successful include powdered neem cake mixed with the top 2-3 inches of sand above a nest (Arun, 2019) or application of a toxic ant bait or pesticide (Hughes, 1971), especially with the active ingredient hydramethylnon (Kelaskar et al., 2016; Smith et al., 2020), to the surface sand above nests. A negative impact of hydramethylnon on non-target arthropods (Plentovich et al., 2010), nest hatching or emergence success (see Miller, 1999 for definitions), or hatching orientation to the sea have not been detected. However, commercial ant bait involving a carrier (such as the cornmeal and soyabean oil in AMDRO™) might attract other predators to the nest and should cautiously be used lest the bait itself increases the risk of nest depredation (Smith et al., 2020).

Birds

Limited records of diurnal (great frigatebird Fregata minor (Lagarde et al., 2001), crow Corvus splendens, brown-headed gull Larus ridibundus, brahminy kite Haliastur indus, (Tripathy & Rajasekhar, 2009), white-bellied sea eagle Haliaeetus leucogaster (Clohessy, 2014), undescribed hawk and kite species (Ti et al., 2011)) and nocturnal (black-necked stork Ephippiorhynchus asiaticus (Whiting & Guinean, 1999)) bird predation on eggs and/or hatchlings are available for the IO & SEA region. Studies worldwide have described different avian predation methods on sea turtle nesting beaches, including the probing of nests for eggs or hatchlings, feeding on eggs exposed by other predators, or preying on emerged hatchlings (Whiting & Guinean, 1999; Burger & Gochfeld, 2014; Korein et al., 2019). Hatchlings are most vulnerable to birds in the short period from when they emerge from the nest and crawl to the sea. ‘Nest-to-surf’ mortality (Erb & Wyneken, 2019) is often low in comparison with mortality due to other predators and does not have a great impact at the population level. However, high rates of predation- such as great frigatebirds
feeding on every hatchling \( n=1,828 \) during all observed nest emergences \( n=38 \) at Europa Island (Lagarde et al., 2001) - may occur at some locations; protection against this level of predation may be neither possible or desirable.

If needed, nest covers made of bamboo or other materials can reduce the access of birds, including different types of vultures, to sea turtle eggs in the nest (Korein et al., 2019), but will have to be checked frequently to ensure hatching release. The presence of human observers during hatching releases should further reduce bird attempts at feeding (Burger & Gochfeld, 2014).

**Cats**

Records of cat (Felis catus) predation on hatchlings are uncommon worldwide, but are included in reports from Qatar (Ficetola, 2008), the Seychelles (Seabrook, 1989), Myanmar (Thi et al., 2011) and Western Australia (Hilmer et al., 2010) in the IO & SEA. Most cats that prey on turtle hatchlings are not domestic pets (categorised as feral, stray, or free-roaming cats in different countries). Turtle nest protection devices against cats have only been used in Qatar where a 1m square plastic mesh square positioned above the nest and buried under 5cm sand significantly reduced predation by both cats and foxes (Ficetola, 2008). The design included a central detachable disc that was removed 1 week before the predicted emergence date to allow hatchlings to escape the nest (see Yerli et al., 1997). Other effective nest enclosures against foxes could also likely reduce cat predation.

Control measures recommended for free-roaming cat populations include removal by trapping (Algar et al., 2011b; Dias et al., 2017), poisoned baiting (Algar et al., 2011b), or hunting (Leo et al., 2018). Several studies suggest combinations of strategies, such as baiting and trapping (Algar et al., 2011b) and removal and neutering (Dias et al., 2017). It is also recommended that pet or house cats be neutered (Algar et al., 2011a) and prevented from roaming free outdoors (Dias et al., 2017) to prevent their contribution to the free-roaming populations of cats.

**Crabs**

Ghost crabs are common on sea turtle nesting beaches worldwide, and predation on eggs and hatchlings in the IO & SEA has been reported from India (Tripathy & Rajasekhar, 2009), Sri Lanka (Ekanayake et al., 2010; Ellepola et al., 2014), the Seychelles (Hitchins et al., 2004), Myanmar (Thi et al., 2011), Malaysia (Ali & Ibrahim, 2002; Chan, 2013), West Papua, and Papua New Guinea (Kinan, 2005). Prey are detected through sight, sensing vibrations, or hearing (Lucrezi & Schlacher, 2014). Only a few eggs per nest are usually preyed by crabs, although they may enter many nests (Korein et al., 2019). Thus, the population-level influence is likely to be low except in specific locations (see Marco et al., 2015) or if crab depredation attracts other predators to the nest (this may increase the likelihood of nest predation by raccoons; Barton & Roth, 2008). An enclosure of small-diameter (0.5cm) mesh around the nest may successfully exclude crabs (Ali & Ibrahim, 2002) but would not allow hatchlings to escape without assistance.

**Crocodiles**

Saltwater crocodiles (Crocodylus porosus) preying on nesting sea turtles, eggs, and hatchlings is only reported from northern and north-western Australia (Whiting & Whiting, 2011). Nest depredation alone occurs in West Papua and Papua New Guinea (Kinan, 2005). Crocodile distribution in the IO & SEA also includes nesting beaches in the Andaman and Nicobar Islands and Southeast Asia, but there have been no similar descriptions from these locations (A. Swaminathan, pers.comm.). There are easier prey species for crocodiles to target so high levels of predation on eggs is unlikely.

**Dogs and Jackals**

Dogs (Canis familiaris) and side-striped (Canis adustus) or unidentified jackals prey on sea turtle eggs and/or hatchlings in South Africa (G. Hughes, pers.comm.), Yemen (Nasher & Al Jumaily, 2015), India (Tripathy & Rajasekhar, 2009), Sri Lanka (Ekanayake et al., 2010; Ellepola et al., 2014), Myanmar (Thi et al., 2011), West Papua, and Papua New Guinea (Kinan, 2005, Hitipieuw et al. 2007). The impact of these canids at a population level has not been reported. Distinguishing between native and introduced species, dogs have previously been shot when posing a serious risk to eggs and/ or hatchlings in South Africa but the indigenous side-striped jackals which posed an equivalent threat were not controlled (G. Hughes, pers.comm.). Habanero pepper powder (Lamarre-DeJesus & Griffin, 2013) or trapping and removal (Eskew, 2012) have reduced turtle nest depredation by coyotes, another canid, and could also be used for dogs and jackals. Similarly, protection devices against foxes would also likely be effective. Semi-domestic dogs (often referred to as ‘village’ dogs in South Asia) should be adequately fed to reduce their likelihood of eating eggs and hatchlings, and access to nesting beaches should be limited by restricting their roaming at night (Ruiz-Izaguirre et al., 2015).

**Foxes**

Descriptions of fox depredation of sea turtle nests in the IO & SEA region are less common than those of other animals. The Rüppell’s fox (Vulpes rueppelli) completely destroyed >80% of unprotected nests on a beach in Qatar.
(Ficetola, 2008), and nests are protected against the European red fox (Vulpes vulpes) in Western Australia (Waayers et al., 2012) and an unidentified species in Pakistan (Waqas et al., 2011). Foxes use olfactory and visual cues to locate turtle nests and learned behaviour about finding nests can be passed from adult to offspring (O’Connor et al., 2017). Adults may raid nests then cache eggs away in a different location to feed young on subsequent nights (Macdonald et al., 1994). In some locations, foxes have been regarded as the single greatest terrestrial predator of eggs and hatchlings (e.g., eastern Australia; Limpus, 2008).

Successful control measures to reduce fox depredation of nests have included combinations of nest enclosures and flags in northeast Brazil (Longo et al., 2009), and trapping with subsequent ethanization, den fumigation with carbon monoxide, and nest enclosures in eastern Australia (O’Connor et al., 2017). Nest enclosures effective against foxes are usually mesh screens. A 1m square piece of plastic mesh, with 100mm openings, held in place above the nest with eight 30cm stakes and covered in 2cm of sand protected loggerhead nests in eastern Australia (O’Connor et al., 2017). Wire mesh, also 1m square, positioned above the nest, buried under 5cm sand, and with a central detachable disc, has been used with loggerhead nests in Turkey (see Yerli et al., 1997). Plastic-covered metal or plastic mesh (1m square; mesh size 70mm) buried 5-10cm below the beach surface protected green, hawksbill, loggerhead and olive ridley turtle nests in Brazil (Marcovaldi & Laurent, 1996; Longo et al., 2009).

**Goannas, Lizards and Monitors (Varanids)**

Varanids, including the Asian or common water monitor (Varanus salvator) in Sri Lanka (Ekanayake et al., 2010), the Andaman and Nicobar Islands (Chandi et al., 2006) and Malaysia (Salleh et al., 2012), the Bengal monitor/common Indian monitor (Varanus bengalensis) in Sri Lanka (Ekanayake et al., 2010), the coastal/yellow-spotted goanna (Varanus panoptes) in northern Australia (Blamires et al., 2003) and an unnamed species in Bangladesh (Islam, 2002), Malaysia (Chan, 2013), West Papua, and Papua New Guinea (Kinan, 2005), depredate sea turtle nests. Depredation rates exceeded 90% of nests on some islands in the Andamans and Nicobars (Chandi et al., 2006). Analysis of goanna scats at Fog Bay in northern Australia indicated that flatback turtle eggs were a major prey item in the dry season (Blamires, 2004) but the impact on the turtle population was unknown.

Varanids likely use visual (e.g., nest mound) and/or chemical (e.g., scent of eggs, fluids or hatchlings) cues to locate eggs (Blamires et al., 2003; Lei & Booth, 2018) and clutches laid on the dune crest may be more vulnerable to depredation than those at dune base (Blamires et al., 2003). Panels of aluminium mesh, described above as effective against foxes (O’Connor et al., 2017), have also reduced high rates of goanna depredation of turtle nests. A top 1m square panel with four side panels of 10-25cm width should be buried to 20cm below the beach surface and the sand replaced to original beach height (Lei & Booth, 2017; Madden Hoff et al., 2019, see also Supplementary Plate 1 for image of nest cover being placed). A single plastic mesh panel- 1.2 x 1.5m, 50mm mesh size, buried to 10cm below the beach surface and secured in place with 40cm wooden stakes at the corners- was effective against yellow-spotted goannas predating on loggerhead turtle nests at Wreck Rock QLD, Australia (Lei & Booth, 2017). However, plastic mesh (90cm x 100cm; mesh size 50mm), buried 10cm below the sand surface and staked with up to nine sand pegs around the perimeter did not reduce goanna depredation of olive ridley turtle nests on the Cape York Peninsula QLD, Australia (Nordberg et al., 2019). “Netlon” mesh (1.5 x 1.5m; mesh size not described but see images) has been used to protect nests against varanids in Malaysia (Chan, 2013).

**Honey Badgers**

Honey badgers (Mellivora capensis) are only found in Africa, West Asia, and the Indian subcontinent. Predation on turtle eggs and hatchlings by this species has only been described in South Africa (Bourjea et al., 2008; de Wet, 2012), Tanzania (West, 2010), and Mozambique; predation is considered low in the first two countries but perceived as an emerging threat and in need of assessment in Mozambique (Williams et al., 2019). Only one study trialled exclusion methods and found cages of wire mesh (90 x 90 x 75cm cage of 5 x 10cm mesh, with the bottom 15cm bent outwards and buried; Boulon Jr., 1999) successfully prevented access of honey badgers to the nest (West, 2010).

**Hyenas**

Hyenas (species not named) or their tracks have been associated with depredated turtle nests in Kenya (Olendo et al., 2016) and India (Tripathy & Rajasekhar, 2009; Karnad, 2017). Enclosures to protect nests from hyenas have not been described, but protection devices used against foxes would probably be effective for the taxon.

**Mongooses**

Water/marsh mongoose (Atilax paludinosus) depredate a low proportion of nests in South Africa (Bourjea et al., 2008; de Wet, 2012) and Sri Lanka (Ekanayake et al., 2010). If excluder devices were needed to protect turtle nests from this species, mesh covers or cages effective against goannas or foxes, as described in the current paper, could be appropriate.
Pigs

Reports of feral or wild pigs depredating turtle nests in the IO & SEA come from Sri Lanka (Ellepola et al., 2014), the Andaman and Nicobar Islands (Chandi et al., 2006), West Papua (previously Irian Jaya; Salm, 1982; Suganuma, 2005; Thebu & Hitipeuw, 2005; Hitipeuiv et al., 2007), and Papua New Guinea (Kinan, 2005). Egg loss due to pig depredation can be severe; for example, ~90% of nests were depredated during some nesting seasons in Sri Lanka (Ellepola et al., 2014). Pigs can quickly become conditioned to recognise sea turtle nests as a food source, and so predator management strategies should be implemented as soon as possible once nest depredation behaviour becomes established in a population (Engeman et al., 2016). Depredation by pigs occurs soon after oviposition, suggesting that visual and olfactory cues are used to find the nest (Whytlaw et al., 2013).

The biological and economic costs of pig destruction of turtle nests was considered to be so high at North Island SC, USA, that a combination of trapping and removal, nighttime sharpshooting, aerial sharpshooting, public hunts, and private hunters with dogs have been used over time to control the animals (Engeman et al., 2019). These methods and their associated cost may not be feasible everywhere. However, enclosures have not been successful at protecting nests from pigs. For example, pieces of plastic mesh (90 x 100cm; mesh size 50mm), buried 10cm below the sand surface and staked with up to nine sand pegs around the perimeter did not reduce pig depredation of olive ridley turtle nests on the Cape York Peninsula QLD, Australia (Nordberg et al., 2019).

Porcupines

The only published record describing porcupines as a predator of sea turtle nest contents comes from Kenya (Olendo et al., 2016). The rate of depredation was unquantified so is likely to be low. If it were to become a problem, nest enclosures described in the current paper as being effective against foxes or goannas would likely be effective.

MONITORING EFFICACY

The number of predator tracks in surface plots at the base of the dune vegetation line and strip transects from dune to shore over fixed time periods can be used to calculate a passive tracking index for predators (see Engeman et al. (2003) for detailed methods and formula) and camera traps can be used to estimate predator abundance (e.g., Gilbert et al., 2020). These methods will also detect population increases among other potential nest predators as the target species is controlled. Nests should be monitored regularly to calculate changes in depredation rates before and after nest protection and/or predator management. The results can guide the choice of strategy, timing, and area for predator control strategies, and also assess the effectiveness of the strategy after it has been implemented. (Engeman et al., 2003, 2005).

SUMMARY AND RECOMMENDATIONS

Depredation of turtle nests play an important role in supporting coastal food chains and nutrient transfer from marine to terrestrial environments, and only requires management if egg and/or hatching loss is likely to threaten population recruitment. While relocation of eggs to hatcheries for protection is a common strategy to reduce the likelihood of predation, other potential strategies include predator removal or exclusion from the nesting beach, deterrents, and nest enclosures. There are clear limitations to physically protecting every nest, especially along beaches on which turtles may nest over tens or hundreds of kilometres. If a known predator is introduced or invasive, then the most efficient method for its population management and/or sea turtle nest protection should be applied. If the predator is a native species, especially long associated with the nesting site, then caution should be exercised and predation control should focus on nest protection rather than predator control, unless the conservation needs of the turtles far outweigh those of the predators.

Finally, researchers are encouraged to consider the potential use of suitable tissues from depredated clutches or hatchlings for studies of sex ratios (e.g., Rebelo et al., 2015; Chabot et al., 2019).

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Maharashtra has a 720km long coastline which harbors a rich marine biodiversity, including the five species of sea turtles known to inhabit the coastal waters of India (Hatkar et al., 2019). However, leatherback turtles (Dermochelys coriacea) have rarely been reported and lack photographic evidence to validate presence of the species. The leatherback turtle is listed as vulnerable by the International Union for Conservation of Nature (IUCN) (Wallace et al., 2013) and is a Schedule-1 species protected under the Wildlife (Protection) Act, 1972 (WPA, 1972) of India. The leatherback turtle is locally known as ‘Kurma’ in Maharashtra (Sanaye & Pawar, 2009). It is highly migratory, spends most of its life offshore, and feeds on scyphozoa (Dodge et al., 2011). Within the wider region, leatherback sea turtles nest at Bird’s Head Peninsula, West Papua (Indonesia), Andaman and Nicobar Islands (India), Godavaya (Sri Lanka) and KwaZulu-Natal (South Africa) (Shanker, 2004).

Very little information is available about the occurrence of the leatherback turtle on the west coast of India. Ten to fifteen years ago, two nests of leatherback turtles were recorded each in Sindhudurg and Raigad districts in the state of Maharashtra (Giri, 2001). However, no authenticated record of the nesting was available. A stranding of a leatherback turtle was recorded from the beach of Devbag, Maharashtra, in 1985 (Karbhari, 1985) but again no photographic record was available of the event. A leatherback turtle (no photos for validation but morphometric measurements support the species identification) was entangled in a gill net off Vizhinjam, Kerala, in 2008 and was released back to the sea (Anil et al., 2009).
and the Fisheries Department of Maharashtra started a compensation scheme in December 2018, under which fishers who cut or otherwise damage their fishing gear to release a marine animal protected under WPA 1972 were given monetary compensation. Several awareness and outreach workshops were carried out in the coastal districts of Maharashtra by the Mangrove Cell to popularise the said scheme and to build a network of fishers to collect secondary data of endangered marine animals. Subsequently, this record of a leatherback sea turtle was shared by a fisher based in Bharadkhol (18.15° N, 72.83° E), a small coastal village in the Raigad district. The turtle was caught in a gill net on 25th May 2018 and was released back to the sea safely by fishers cutting the net. This is the first photographic record (Figure 1) of a leatherback sea turtle from Maharashtra. Using ImageJ software, the estimated length of the turtle was ~1.2m.

More needs to be known about the occurrence of leatherback sea turtles off the Maharashtra coastline, and could be gathered from similar reports from fishers or a research study.

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SILKY SHARK FEEDING ON A JUVENILE GREEN TURTLE IN OFFSHORE WATERS OF PAKISTAN, NORTHERN ARABIAN SEA

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Turtles play an important role in coastal and offshore ecosystems as they feed on organisms such as jellyfish, crustaceans, mollusks, and seaweed (Bjorndal, 1997). They, on the other hand, are preyed upon by a number of animals and form an important part of their diet (Hirth, 1971; Stancyk, 1981; Witzell, 1987). Sharks are among the major predators of sea turtles (Cabrera-Chavez-Costa et al., 2010; Hammerschlag et al., 2015; Estupiñán-Montaño et al., 2018). For example, Heithaus (2001) and Simpfendorfer et al. (2001) found that turtles are an important component in the diet of tiger sharks (Galeocerdo cuvier). However, less information has been published on the diet or feeding habits of the silky shark (Carcharius falciformis). Cabrera-Chavez-Costa et al. (2010) reported that silky sharks in the Gulf of Mexico (Atlantic Ocean) mainly preyed on teleost fish and in the Gulf of Tehuantepec (Pacific Ocean) on the crab Portunus xanthusii affinis. Galvan et al. (1989)
found that silky sharks from the Gulf of California mainly fed on the red crab (*Pleuroncodes planipes*) and to a lesser extent on fish and benthic crustaceans in Baja California. This paper describes an observation of a silky shark feeding on a juvenile green turtle (*Chelonia mydas*) in Pakistan (Northern Arabian Sea).

Fishers caught a 1.35m long female silky shark on the 1st January 2017, approximately 65km south of Ormara, Balochistan. The authors observed the shark in the Karachi Fish Harbour on the 4th January and found its distended stomach to be of interest. The authors dissected the specimen, removed the stomach, and took it to a laboratory at the Marine Fisheries Department, Karachi.

Eight undigested fragments (Figure 1) of tissue and carapace of what appeared to be an individual juvenile green turtle were found. No other material was present in the stomach, and the remainder of the turtle may have been lost or eaten by another carnivore. The flesh and carapace scutes of the juvenile green turtle had retained their original colour, suggesting that the digestion process was in an early stage. The upper and lower jaws of the turtle were intact and measured 2.2cm in length (Figure 1). Based on the relationship between body length and mouth width from other silky shark specimens, the shark mouth was estimated as 10.7cm wide. These measurements indicate that the 1.35m long silky shark would not have been able to swallow the green turtle in a single gulp. The eight fragments indicate that the shark must have ingested the turtle in multiple bites during a series of lunges.

The silky shark was caught in the offshore waters of Pakistan (Northern Arabian Sea) where the water depth was ~700m. Small green turtles probably inhabit these waters during the oceanic juvenile stage of their life cycle (Bolten, 2003). The capture site was ~65km south of Ormara, a green turtle nesting beach (Groombridge, 1989), so there is the potential for interactions between silky sharks and breeding turtles, hatchlings, and/or turtles of other life stages feeding in the Arabian Sea. Since this report is based on a single observation of a silky shark feeding on a juvenile green turtle, we recommend closer inspection of other shark specimens at harbours to increase the understanding of shark predation and other in-water threats to turtles at different life stages and in different habitats.

**Literature cited:**


The 8th Meeting of the Signatories (MOS8) to the IOSEA Marine Turtle MOU was held from 21 to 24th October 2019 in Da Nang, hosted by the Government of Viet Nam. Representatives of 25 Signatory States and two non-Signatory Range States, as well as several Advisory Committee Members and many observers, attended the meeting. The Advisory Committee met just before MOS8, from 16 to 18th October, with the days in between used to finalise its advice regarding priorities and the future direction of the MOU.

The main output of MOD8 is a Work Programme (available at www.cms.int/iosea-turtles/en/meeting/MOS8) for the years 2020-2024, which prioritises recommended activities for the Signatory States, the Advisory Committee, and the Secretariat. It was developed based on a review of all recommendations of past Meetings of Signatories and other intersessional meetings, as well as recommendations made at MOS8.

Twenty-six Signatory States submitted national reports before the meeting, providing information on the state of implementation of the MOU in their countries, as well as problems faced by marine turtles in their territories. Countries also had an opportunity to indicate where they saw priorities and capacity-building needs both nationally and in their sub-region. MOS8 also set up a working group, which will review the national reporting format to ensure it is up to date and fulfils its function.

The meeting also discussed guidelines for the review of environmental impact assessments (EIAs) of developments with an impact on marine turtles and turtle habitat. It was recognised that a balance had to be struck between economic development and nature conservation. Oil and gas installations, harbour facilities, urban development and golf courses all have potential effects on natural habitats. EIA processes follow a basic pattern in all countries, but any flaws in the process can render them ineffective. The presence of endangered species should be an automatic trigger for EIA, but the complex life cycle of marine turtles makes it difficult to assess the effects of a project on the taxa. The document presented to the meeting includes a table setting out safe distances and suitable buffer zones. Specifically relating to marine noise, guidelines adopted by CMS were also made available to Signatories.

Other topics on the agenda included: illegal take and trade of marine turtles, under which both the relevant outcomes of a study conducted by CITES and a study investigating this issue in the Solomon Islands were discussed; opportunities for collaboration with the Convention on Migratory Species (CMS) and other international organisations, such as CITES, Ramsar and Regional Fisheries Management Organizations (RFMOs); and guidance on beach management and hatcheries.

Amongst the highlights of the meeting was the inclusion of Con Dao National Park, Viet Nam, into the IOSEA Network of Sites of Importance for Marine Turtles in the Indian Ocean. The IOSEA Site Network aims to promote and coordinate the long-term conservation of sites of regional and global importance to marine turtles and their habitats. When the Network was launched in 2014, 10 sites were evaluated according to a set of defined criteria and accepted. Con Dao National Park has become the 11th Site in the Network (information on all sites can be found at www.cms.int/iosea-turtles/en/activities/site-network).

Con Dao National Park is situated off the south-east coast of Viet Nam and consists of 14 islands. The smaller islands surrounding Con Son Island especially offer important nesting beaches. The waters of Con Dao provide feeding habitats for green (Chelonia mydas) and hawksbill (Eretmochelys imbricata) turtles, both of which are included on the Red List of the International Union for Conservation of Nature (IUCN). Viet Nam introduced legal protection of marine turtles in 1987, when consumption of meat and eggs became illegal. The first two marine turtle conservation stations in Con Dao were established in 1989, and a further three stations in 1996. Thanks to the conservation efforts of the Vietnamese Government, there has been a dramatic decline in poaching of marine turtles and egg collection. Con Dao was established as a National Park in 1993. However,
activities in the areas surrounding the site are causing increasing pressures to the habitats. These pressures include: illegal take of marine turtles and collection of their eggs for trade; marine pollution from untreated waste; and overfishing and destructive fishing practices, which disturb the foraging and nesting of marine turtles. To enhance the protection of the marine turtle habitats, the IOSEA Site Network aims to address these threats. Further information about the IOSEA MOU can be found online (www.cms.int/iosea-turtles/en/). Pictures from the meeting can be found in the related news item on the IOSEA website (https://www.cms.int/iosea-turtles/en/news/signatories-iosea-marine-turtle-mou-agree-new-work-programme-0).

For any questions regarding the IOSEA Marine Turtle MOU, please contact the Coordinator, Ms Heidrun Frisch-Nwakanma. The official email address, iosea@un.org, is also again operational.

**IMPORTANT TERMS EXPLAINED**

**IOSEA Marine Turtle MOU:** The Memorandum of Understanding on the Conservation and Management of Marine Turtles and their Habitats of the Indian Ocean and South-East Asia is a framework through which States, territories, inter- and non-governmental stakeholders of the region, as well as other concerned States, can work together to conserve marine turtle populations and their habitats. The objective of the MOU is to protect, conserve, replenish and recover marine turtles and their habitats, based on the best scientific evidence, taking into account the environmental, socio-economic and cultural characteristics of the Signatory States. It came into effect in 2001. A Conservation and Management Plan is part of the MOU. (more information at www.cms.int/iosea-turtles/en)

**Signatory States:** Thirty-five countries have so far signed the MOU (see list at www.cms.int/iosea-turtles/en/about/membership)

**non-Signatory Range States:** Countries in the region (or active in the region) that have not yet signed the MOU, but are invited to do so.

**Advisory Committee (AC):** In order to help them achieve the MOU’s objective, Signatories established an AC, which provides scientific, technical and legal advice to the Signatories, individually and collectively. The AC currently consists of ten members representing a range of relevant expertise (for details, see www.cms.int/iosea-turtles/en/organizational-structure/advisory-committee).

**Secretariat:** Based in Bonn, Germany, it acts as the coordinating body of the MOU and was established to assist communication, encourage reporting and facilitate activities between and among Signatory States, sub-regional institutions and other interested States and organizations (see also www.cms.int/iosea-turtles/en/organizational-structure/secretariat-iosea).

**Meeting of Signatory States (MOS):** It is the decision-making body of the MOU. The MOS meets regularly subject to capacity, need and availability of funding, to review progress made and difficulties encountered in the implementation of the MOU and to lay down the priorities for the next years. Meetings of the MOS are open to observers, such as researchers and NGO representatives (for more information on meetings held so far, go to www.cms.int/iosea-turtles/en/about/iosea-organisational-structure).
Artificial light is increasing globally by around 2% per year (Kyba et al., 2017) and it is recognised as an emerging conservation issue for wildlife (Russart & Nelson, 2017). Hatchling marine turtles are vulnerable to light pollution as it disrupts natural light cues used for finding the ocean (Witherington & Martin, 2003) and dispersing through the nearshore environment (Thums et al., 2016; Wilson et al., 2018). Artificial light may also deter adult marine turtles from nesting on lit beaches (Salmon, 2003). Increases in artificial light have been observed to affect marine turtles globally (Lutcavage et al., 2017), and artificial light has been identified as a threat to marine turtles in the Indian Ocean region (Karnad et al., 2009; Kamrowski et al., 2012, Department of Biodiversity Conservation and Attractions, 2017; Chalastani et al., 2020).

To address this conservation challenge, the Australian Department of Agriculture, Water and the Environment in collaboration with the Western Australian Department of Biodiversity, Conservation and Attractions has developed National Light Pollution Guidelines for Wildlife including Marine Turtles, Seabirds and Migratory Shorebirds (Commonwealth of Australia, 2020). The Light Pollution Guidelines aim to raise awareness of the potential impacts of artificial light on marine turtles and provide a framework for assessing and managing these impacts near important nesting beaches. This framework provides foundational knowledge on the potential biological impacts of artificial light, as well as consistent, standardised and transparent processes and expectations for assessing, measuring, auditing and managing artificial light around wildlife.

**BEST PRACTICE LIGHT DESIGN PRINCIPLES**

The Light Pollution Guidelines advocate a best practice approach to managing artificial light for wildlife, including reducing sky glow. To reduce sky glow these best practice principles (Figure 1) should be implemented for all outdoor lighting:

1. Start with natural darkness. Identify the reason for artificial light and only add light for specific purposes.
2. Use adaptive, smart controls for lighting. Advances in technology mean that lighting can be controlled by timers, motion sensors, and automatic dimmers.
3. Avoid light spill. Only light the intended area or objects. Keep lights close to, and oriented towards the ground as upward light contributes to sky glow.
4. Use the lowest intensity lighting appropriate to the task. Lighting should be the lowest intensity needed to illuminate the area of interest.
5. Use non-reflective surfaces. Reflected light can contribute to sky glow.
6. Use lights with little or no blue wavelengths. Shorter wavelength (blue) light refracts more as it travels through the atmosphere, contributing more to sky glow than longer wavelength (yellow, orange and red) light. Also, turtle hatchlings are more sensitive to blue-green wavelengths than orange-red.

**ENVIRONMENTAL IMPACT ASSESSMENT FOR ARTIFICIAL LIGHT**

Around important marine turtle nesting beaches, the Light Pollution Guidelines recommend an Environmental Impact Assessment (EIA) approach. This should include consideration of the biological management objectives regarding artificial light, the proposed lighting design and mitigation and a risk assessment including the likelihood and consequence to nesting marine turtles or hatchlings. The effectiveness of mitigation should always be reviewed using biological monitoring and light auditing and by employing an adaptive management approach.

**HUMAN-WILDLIFE LIGHTING CHALLENGES**

The Light Pollution Guidelines recognise the sometimes-
conflicting requirements for human safety and wildlife conservation and do not seek to inhibit the benefits afforded by artificial lighting for humans. Instead the Guidelines note the need to find creative solutions to address both wildlife conservation needs and human safety requirements.

**APPLICABILITY OF GUIDELINES**

The Light Pollution Guidelines provide specific advice for management of light near important habitat for marine turtles, seabirds, and migratory shorebirds, however the management approach is broadly applicable for any wildlife for which there is evidence that artificial light has an adverse impact. The Guidelines note that artificial light has the potential to impact on a broad range of threatened and migratory species. It is also recognised that incorporating best practice lighting design into all infrastructure will not only have benefits for wildlife, but the environment more broadly through reduced energy consumption. This will in turn provide economic benefit for light owners and managers.

Although the Light Pollution Guidelines were developed within the Australian context, the pervasive nature of light pollution means that the broad principles, process, and technical information provided in the Light Pollution Guidelines can be applied in other countries experiencing similar challenges. On this basis, the Australian Light Pollution Guidelines were presented to 13th Conference of the Parties to the *Convention on the Conservation of Migratory Species of Wild Animals* (CMS) in Gandhinagar, India, in February 2020. The Guidelines were endorsed, and the Secretariat requested to promote the Light Pollution Guidelines amongst subsidiary agreements to the CMS, such as the Indian Ocean South East Asian Marine Turtle Memorandum of Understanding (IOSEA).

Australia presented the draft Light Pollution Guidelines at the IOSEA Meeting in Da Nang City, Viet Nam in October 2019 (see Frisch-Nwakanma (2020) in this issue) where the Signatory States supported the Guidelines and agreed to consider applying the Guidelines within national jurisdictions.


**Literature cited:**


Commonwealth of Australia. 2020. National Light Pollution Guidelines for Wildlife including Marine Turtles, Seabirds and...


Frisch-Nwakanma, H. 2020. 8th meeting of the Signatories to IOSEA Marine Turtle MoU agrees on new actions to protect turtles. Indian Ocean Turtle Newsletter 32: 45-46.


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