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Sea Turtle Conservation: Reviewing the Efficacy of Land- and Sea-based Management Strategies for Loggerhead (Caretta caretta) and Leatherback (Dermochelys coriacea) Sea Turtles

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And You. Because you're awesome.

Table of Contents

Abstract

As cosmopolitan species, loggerhead and leatherback turtles are confronted with a multitude of threats as they progress through their respective life stages. These range from depredation and poaching of eggs, hatchlings, and females on nesting beaches, to incidental hooking in pelagic longline fisheries and capture in trawl fisheries. Some threats are species specific on regional scales, though most impact both species. To confront these threats, various conservation strategies have been developed and implemented, including monitoring and caging of nests and changes to hook shape and trawl design. Here, current conservation methods are presented and discussed on a global scale for both species. Population modeling was employed to elucidate the impacts these strategies are having for loggerhead turtles in the North Atlantic. Unfortunately, even with the myriad of strategies employed throughout the world, most populations of these species are still declining. This arises due to a poor understanding of several of the fundamental elements of population dynamics for each species, deficient tracking of fisheries impacts, and a lack of unified conservation plans to address population declines on regional and global scales.

Keywords: loggerhead, leatherback, conservation strategy, nesting beach conservation, nest depredation, fisheries management, pelagic longline fishery, circle hook, mackerel bait, passive net fishery, net height, trawl fishery, turtle excluder device, population modeling, population matrix, projection matrix, age-classified, age-based, stage-classified, stage-based, remigration, survivorship, survival rate, stage duration

Introduction

All seven extant sea turtle species have undergone drastic reductions in population sizes worldwide and are listed as endangered or critically endangered by the International Union for Conservation of Nature (IUCN; www.iucnredlist.org). The degree to which each species is endangered varies by population, with some populations showing indications of recovery (e.g., green turtles, *Chelonia mydas*, in Hawaii; Balazs and Chaloupka 2004) while other populations are still rapidly declining (e.g., leatherback turtles, *Dermochelys coriacea*, in the Mexican Pacific; Martinez, et al. 2007). In recent years, many conservation strategies have been employed in an attempt to better manage and protect these species. Unfortunately, turtles are still affected by a myriad of threats, including habitat degradation (Witherington, Hirama and Mosier 2011), pollution (J. G. Derraik 2002), overfishing and harvesting of eggs (Wilson and Tisdell 2001, Martinez, et al. 2007), direct interactions with humans on nesting beaches and at sea (Herrera-Silveira, et al. 2010), and incidental catch in fisheries (Wallace, et al. 2011). Globally, conservation and management initiatives to combat threats to sea turtle populations have been implemented (Dutton, et al. 2005, Engeman, et al. 2003). Among these, nesting beach conservation (i.e., the protection and monitoring of nesting beaches; Bjorndal, et al. 1999, Garcia, Ceballos and Adaya 2003) and fisheries management (the development and use of modified fishing techniques and gear; Arendt, et al. 2012, Price and Gearhart 2011) offer varying degrees of by mitigating the effects of land- and sea-based threats (Dryden, et al. 2008, Green, et al. 2009) and are at the forefront of marine turtle conservation. Other conservation strategies employed for different purposes (e.g., coral reef marine protected areas, seagrass restoration) that, though do not specifically target turtles, can affect them by protecting and restoring foraging grounds (Pressey and Bottrill 2009).

Marine turtles are large bodied, highly mobile organisms with complex life history patterns that present a number of challenges to population level conservation and management (Gruss, et al. 2011). Though each marine turtle species exhibits different life histories, all have a surface-pelagic juvenile life stage (Wallace, et al. 2010). This, alone, poses a significant challenge in designing comprehensive management strategies (Wallace, et al. 2011). It has long been believed that during these pelagic “lost years”, post-hatchling turtles were passive drifters that moved around the major ocean basins along boundary and other major ocean currents (Carr 1987a). However, recent satellite tracking evidence revealed that young turtles may actively seek out suitable pelagic habitats (Mansfield, Saba and Musick 2009, Putman and Mansfield 2015). This vagrant lifestyle brings juvenile turtles into the territorial waters of numerous nation-states and results in interactions with fishing gear in these highly productive habitats.

All turtles spend time in terrestrial zones as eggs and early hatchlings on nesting beaches, with females returning at varied intervals to nest. Immediately after hatching, hatchlings orient and crawl toward the ocean and began the hatchling swim frenzy stage for the next 24 hours to several days (Wyneken and Salmon 1992, J. Spotila 2004). The remaining life stages differ among species, with notable variations between loggerheads and leatherbacks. Loggerheads will spend 7-10 years in the juvenile oceanic stage, slowly transitioning to deeper, benthic foraging habitats as they grow older and larger (J. Spotila 2004, Heppell, et al. 2003, Heppell, Snover and Crowder, 2003). As subadults and adults, loggerhead turtles will recruit to neritic habitats, and generally limit their time in the pelagic environment to periods of migration between feeding grounds, mating grounds, and nesting beaches, though a small portion of Atlantic and Pacific populations will return to the oceanic zone as large juveniles and adults (Bolten 2003). In contrast, leatherback turtles will spend nearly their entire lives in the oceanic zone, returning to neritic zones to breed and nest, or for brief visits during their continuous migrations in pursuit of jellyfish and other soft-bodied animals. As they grow, leatherbacks will expand their foraging depth range, eventually able to dive as deep as 1230 meters as adults, though the majority of their time is spent above 200 m (J. Spotila 2004, Hays, Houghton, et al. 2004).

The varied life histories of sea turtles complicate conservation attempts. In many cases, effective management strategies must be tailored to specific stages of a species life history (Gruss, et al. 2011). For example, due to the long distances traveled during the oceanic juvenile life stages, management and conservation measures are most effectively addressed using international treaties and conventions. However, regulations put forth from these agreements are difficult to enforce due to the expanse of the oceanic environment, the need for cooperation among various sovereign states, and other compounding factors (Wold 2002). Much of the difficulty is attributed to the disparity between the implicit governing rights in different areas of the world's oceans. That is, each state possesses sovereign rights over the natural resources that reside within its exclusive economic zone, which extends up to 200 nautical miles from the coast of each state. Conversely, on the high seas outside of these zones, no state maintains sovereign rights, and thus all resources may be exploited by all states. Specifically, exploitation may only occur in a way that benefits all states, and that resources must be conserved (Wold 2002). While these rules provide the authority for states to regulate threats to sea turtles, the inherent ambiguity provides no regulatory framework or guidelines towards this end, thus affording turtles, and other wide-ranging species, inadequate protection.

There are international agreements that have significantly curbed past threats to sea turtles, namely, the Convention on International Trade in Endangered Species of Wild Fauna and Flora (CITES) and the Convention on the Conservation of Migratory Species of Wild Animals of 1979 (CMS). CITES has been largely responsible for dealing with overharvesting for consumption and commercial use, which was the major precursor to the modern threats sea turtles face today. This banned the trade of turtle products for all purposes, which removed the market for turtle fisheries, effectively preventing active legal taking of eggs and adults in most areas. Additionally, many legal instruments broadly address responsible fisheries practices, such as the United Nations Convention on the Law of the Sea, implemented in 1994; the 1995 United Nations Agreement on Straddling Fish Stocks and Highly Migratory Fish Stocks; and the 1995 FAO Code of Conduct for Responsible Fisheries. Unfortunately, these instruments exhibit many of the ambiguities mentioned above.

While CITES has been effective in reducing the commercial trade of sea turtles and their parts, there remains no binding international policy, treaty, or convention that addresses land- and sea-based threats for sea turtles (Wold 2002). Instead, many regional fisheries bodies (RFB; i.e., any organization that oversees a defined fishery) have voluntarily incorporated bycatch reduction strategies, including those for sea turtles. Among these are the five Tuna Regional Fishery Management Organizations (RFMO), which include the Inter-American Tropical Tuna Commission (IATTC) and the International Commission for the conservation of Atlantic Tunas (ICCAT) that operate in the eastern Pacific and Atlantic, respectively (Coelho, Fernandez-Carvalho and Santos 2013); along with several other RMFOs who oversee non-tuna fisheries in world's oceans. With the lack of international agreements, RFBs have implemented guidelines from other sources, such as the Food and Agriculture Organization of the United Nations (FAO).

In recent years, the FAO has issued specific guidelines to reduce sea turtle bycatch. The most recent version of these guidelines outlines gear recommendations, fisheries management strategies, and the handling and release of incidentally captured turtles (FAO Fisheries and Aquaculture Department 2009). In 2007, the IATTC implemented the FAO guidelines for turtle bycatch mitigation; including lowering mortality of caught turtles and reducing injuries during release (Benaka, Cimo and Jenkins 2012). Additionally, the tuna RFMOs and RFBs commonly follow the guidance of the Inter-American Convention for the protection and Conservation of Sea Turtles (IAC). The IAC calls for measures barring the intentional taking of turtles, compliance with CITES, preventing habitat degradation on nesting beaches and in the water, encouraging research into turtle conservation, providing outreach and education to the public and stakeholders, and mitigating bycatch by modifying gear and using Turtle Excluder Devices (TEDs) (Inter-American Convention for the Protection and Conservation of Sea Turtles 2015).

Many countries have introduced regulations to monitor turtle conservation within their own territories, including the United States, Australia, Brazil, Japan, and Portugal. These regulations help address bycatch on vessels that fly their flags, but also serve to organize the other major realm of sea turtle management: nesting beach conservation. In the United States, the active bycatch mitigation and nesting beach conservation strategies grew from the Endangered Species Act (ESA, 1973). The ESA lists all species of sea turtle, except the flatback, as threatened or endangered, thus mandating their conservation in national waters and on nesting beaches following the recovery plans written for each species in 1990 and 1991. Other countries have similar legal instruments, including Costa Rica (Ley de Conservacion de la Vida Silvestre 1992), and the multi-nation effort in the Mediterranean (Action Plan for the Conservation of Mediterranean Marine Turtles 1989).

To ensure that relevant conservation measures are implemented, RFBs and national governments alike have funded research into best practices for both nesting beach conservation and bycatch mitigation, though the latter is far better funded, especially in regard to longline research (Lewison and Crowder 2007). The Inter-American Tropical Tuna Commission (IATTC), which includes the governments of twenty-five countries throughout the Americas, Europe, Asia, and the Pacific Island Nations, regulates conservation and management for tuna fisheries in the eastern Pacific Ocean. In 2007, the IATTC began actively conducting research investigating the impacts hook type, bait type, and fishing gear set depth, have on sea turtle and target species catch rates. Additionally, the IATTC implemented requirements for additional fisheries observers and mandates that all vessels carry equipment for the de-hooking and release of entangled turtles (IATTC, Resolution to mitigate the impact of tuna fishing vessels on sea turtles 2007). The role of these observers is to collect data on gear choice, fishery methods, and information on turtle-fishing gear interactions. Other groups, such as the Eastern Pacific Regional Sea Turtle Bycatch Program (Andraka, et al. 2013) and SELECT-PAL (Redução das capturas acessórias na pescaria de palangre desuperfície) in the South Atlantic (Santos, et al. 2013), are pursuing similar research and regulations.

Bycatch mitigation measures differ significantly between fisheries due to variations in oceanographic conditions (e.g., depth, bathymetrics, current features), target catch, gear choices (e.g., longlines, trawl nets, seine nets), and geopolitical influences. For longlines, the most common measures are varying gear type (e.g., J-hooks to circle hooks), varying bait (e.g., squid to mackerel), and limiting temporal and geographic access to fisheries (O'Keefe, Cadrin and Stokesbury 2013, Amorim, et al. 2014). In trawl fisheries, Turtle Excluder Devices or Trawling Efficiency Devices (TEDs) are often used in fisheries that interact with sea turtles (Jenkins 2012). Other fisheries, such as pound net fisheries, also use gear modifications (e.g., net height modifications) to address sea turtle bycatch.

Compared to the challenges of protecting turtles on the high sea, nesting beach conservation can often be addressed within the jurisdiction of a single country, (e.g., the United States, Australia, Brazil). In the United States, the ESA (1973) mandates the protection of sea turtles on the beaches, while the Action Plan for the Conservation of

Mediterranean Marine Turtles (1989) dictates the same protection for all 21 nation states surrounding the Mediterranean, as well as the European Community. In areas where nesting beach conservation is difficult to support due to funding issues, governmental instability, or other compounding factors, international conservation groups can aid in providing structure and guidance for beach management programs. The Wider Caribbean Sea Turtle Conservation Network (WIDECAST) is one such organization who supports nesting beach conservation, and other conservation initiatives, in over 40 countries in the Caribbean (Eckert 2005).

Management programs differ from beach to beach, but common measures include protecting eggs from predation, both natural and anthropogenic, with the use of cages, predator removal, and effective monitoring (Kornaraki, et al. 2006, Martinez, et al. 2007). Additionally, programs that mitigate disorientation of nesting females and hatchlings by controlling beach lighting (Witherington and Martin 2000), and regulate development of the beach to maintain habitat integrity (Witherington, Hirama and Mosier 2011) are tailored to fit the needs of each beach.

With the exception of natural nest depredation from terrestrial predators (which is limited due to nest depth), nesting beach protective measures are uniformly beneficial to both species. Protecting these crucial life stages (i.e., eggs, hatchlings, and nesting mothers) is a key step in maintaining sustainable populations. Conversely, different fisheries interact with the two species to differing degrees and at different life stages. Pelagic fisheries primarily affect loggerheads during their oceanic life stages, while leatherback turtles spend the majority of their lives in the pelagic environment, even as adults, and are exposed to these fisheries for more of their lifetime (Luschi, Hays and Papi 2003).

Though protecting the initial life stages is important, previous population models for loggerheads in the North Atlantic found that survivorship during the juvenile life stages had the largest effect on long-term population trends (Heppell, Crowder, et al. 2003). Thus, effectively managing fisheries to reduce their impacts on these sensitive life stages may be a larger priority than nesting beach conservation. Unfortunately, the survivorship and duration values used for these age classes are based on extrapolated estimates derived from life history tables, tagging studies, and other sources. Using assumptions gathered during this review, along with updated estimates for survivorship, life stage duration, and age at sexual maturity, I adapted and updated the loggerhead population model. In doing so, I reaffirmed the assumptions from previous modeling exercises and created a new model structure that more accurately compensates for the variable nesting trends of the adult life stages. Estimation of the variables involved in this modeling requires intense, long-term studies that are lacking in the majority of sea turtle populations. Thus, my models will similarly focus on the North Atlantic loggerhead population, as values for annual survival rates, stage duration, and age at sexual maturity are not well established for leatherbacks (Turtle Expert Working Group 2007).

Species Profiles

Loggerhead Turtles

Loggerhead turtles are a globally distributed species, with 10 subpopulations recognized by the IUCN (listed in Figure 1 as regional management units; Casale and Tucker 2015). Loggerheads begin their lives on beaches spread throughout the tropical and subtropical Atlantic, Pacific, and Indian Oceans, as well as the Mediterranean Sea, with the two largest nesting populations occurring along the East coast of Florida and Oman (J. Spotila 2004). After emerging from the nest and reaching the water, hatchlings spend the next few days in a “frenzy” and “postfrenzy” swim (Wyneken and Salmon

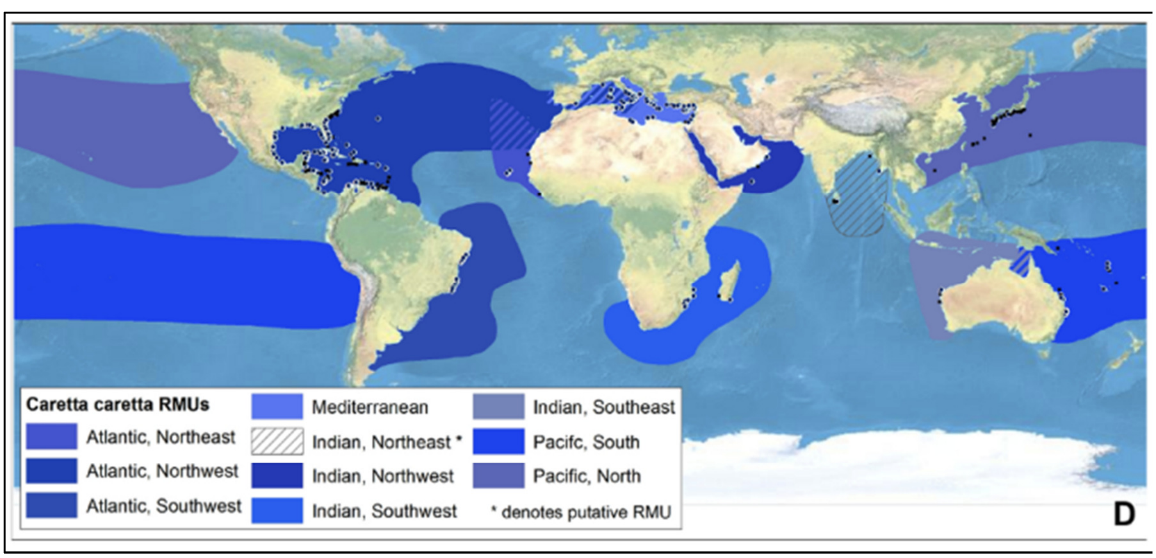

Figure 1 Global loggerhead range highlighting Regional Management Units and nesting sites. RMUs were identified as being geographically and genetically distinct populations. (Adapted from Wallace, et al. 2010.)

1992) oriented straight toward the nearest major ocean current. If an individual successfully evades the pitfalls that hatchling sea turtles face (e.g., terrestrial and marine predation, dehydration, exhaustion) they will be swept into the oceanic zone. Here, they seek out and ride major boundary currents and ocean gyres, e.g., the Gulf Stream-Azores system in the North Atlantic or Japan Current in the North Pacific (J. Spotila 2004, Putman, Bane and Lohmann 2010). In these systems, they find shelter and food in convergence zones, like those found along the boundaries of fronts and eddies (Carr 1987a, Polovina, Kobayashi, et al. 2000).

Post-hatchling loggerheads remain in the oceanic environment for 6.5-11.5 years, often referred to as the "lost years" (Bolten 2003a), with occasional stops at oceanic island chains, e.g., the Azores and Cape Verde in the Atlantic (Bolten, Bjorndal and Martins, et al. 1998). In the Pacific, hatchling and juvenile loggerheads spend time along a boundary region in the North Pacific known as the transition zone chlorophyll front (Polovina, Howell, et al. 2001, Kobayashi, et al. 2008). After reaching an average curved carapace length of 46-64 cm (depending on population), Atlantic loggerheads enter their sub-adult stage and recruit to neritic habitats where they will remain as they continue to develop into adults (Bolten 2003b).

Adult Atlantic loggerheads spend the majority of their time in neritic zone foraging habitats largely comprised of mud and hard bottom areas such as bays, channels, and sounds, along with reefs and oil platforms (J. Spotila 2004). Loggerheads in tropical waters typically show little temporal variation in foraging sites (Rees, et al. 2010), while those in temperate waters may range hundreds of kilometers (J. Spotila 2004). A female loggerhead reaches sexual maturity at 17-33 years of age and an average carapace length of 92-103 cm (depending on population) with reproduction starting towards the end of this range for loggerheads in the North Atlantic (J. Spotila, 2004). The exact start of nesting season (females laying eggs on the beach) varies for loggerheads, but most occur during the summer months in each respective hemisphere (J. Spotila 2004).

Depending on the extent of a population's foraging range and its proximity to breeding habitats (adjacent to nesting beaches), loggerheads may travel through the oceanic zone to mate (Bolten 2003b). Loggerheads show high nesting site fidelity, returning to the same beach they hatched on even if they share foraging grounds with other nesting populations (J. Spotila 2004). Loggerheads nest every 2-4 years, averaging 3.9 clutches of roughly 112 eggs with an inter-nesting interval of 12-17 days (J. Spotila 2004). The nests of loggerheads are shallow, compared to those of larger species, and are simple to locate, making them easy targets for depredation and poaching on beaches where nests are poorly protected.

The diets of loggerheads change drastically between the oceanic (hatchling and juvenile) and neritic (juvenile, sub-adult, and adult) life stages (Bolten 2003b). In oceanic habitats, loggerheads feed at the surface on a variety of flora and fauna that gather at convergence zones, including sargassum, crab zoea, fish eggs, and barnacles (Bolten 2003b, J. Spotila 2004). Once juveniles enter the neritic zone, they experience an ontogenetic shift and transition to benthic prey, particularly hard-shelled slow or sessile organisms such as crabs, conches, and mussels (oceanic juveniles may also demonstrate this prey selection during stopovers at seamounts and islands in waters less than 650 m; Bolten 2003b, J. Spotila 2004). By traversing such a wide variation in habitats, loggerheads encounter a varied assortment of fishing gear, from long lines as pelagic juveniles, to trawl fisheries in neritic habitats as adults.

Leatherback Turtles

The largest of the sea turtles, leatherbacks also exhibit the widest geographical range of any extant turtle species, venturing into the cold waters of the North Atlantic and around the southern tip of Africa during their trans-oceanic migrations (J. Spotila 2004, Nel 2012). The IUCN recognizes seven distinct subpopulations of leatherbacks spread throughout their cosmopolitan distribution (listed in Figure 2 as regional management units; Wallace, Tiwari and Girondot 2013). The Northeast Atlantic leatherback population has demonstrated moderate resilience to the threats that face it and is categorized as a population of "Least Concern" by the IUCN. The two other Atlantic populations, along with those in the Indian and Pacific Oceans, are all considered "Data Deficient" or "Critically Endangered" (Wallace, Tiwari and Girondot 2013).

Like other sea turtles, their nesting beaches are largely limited to the tropics and sub-tropics adjacent to powerful ocean current systems. Contrary to most turtle species, though, leatherbacks do not exhibit high site fidelity to specific beaches. Instead, nesting leatherbacks form large nesting populations across multiple beaches within nesting

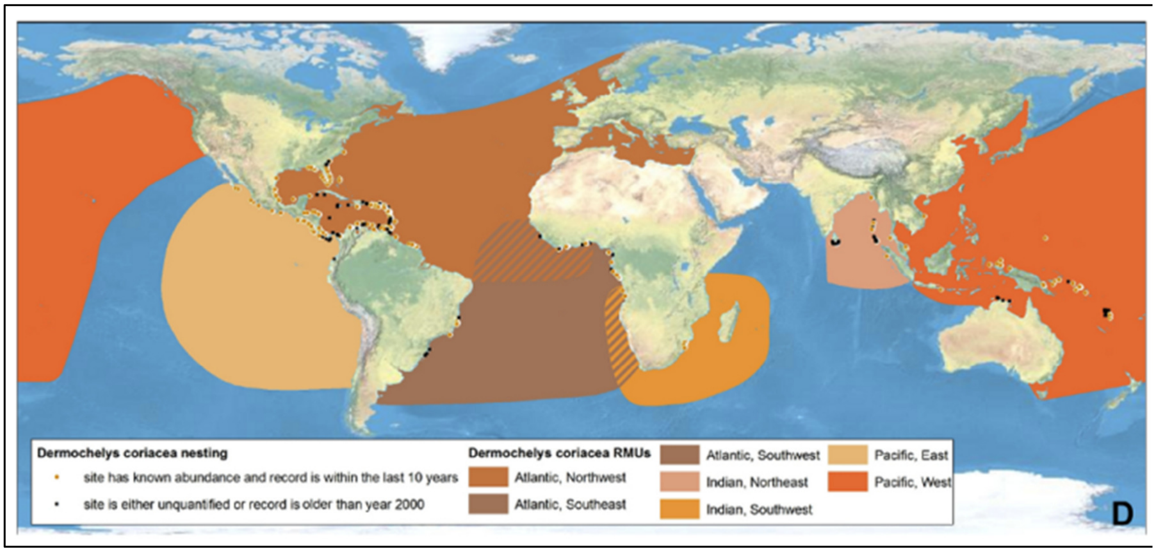

Figure 2 Global leatherback range highlighting Regional Management Units and nesting sites. RMUs were identified as being geographically and genetically distinct populations. (Adapted from Wallace, et al. 2010.)

regions; the largest is the Western Atlantic nesting population in French Guiana and Suriname, possibly including Trinidad and Guyana (Dutton, et al. 1999, Girondot, et al. 2007). Leatherbacks nest throughout the Caribbean, in Gabon in the eastern Atlantic, in Indonesia and Japan in the western Pacific, along the Pacific coasts of Costa Rica and Mexico, and a couple spots in the Indian Ocean (J. Spotila 2004, Girondot, et al. 2007, Patino-Martinez, et al. 2008). In the western Pacific, nesting occurs nearly year-round due to the existence of two distinct nesting sub-populations: arboreal summer and arboreal winter. These arise from foraging populations in the North Pacific and South Pacific, respectively (Benson et al. 2007). These large-scale dispersion patterns make targeted management strategies difficult once the turtles leave the breeding grounds. Additionally, many of the beaches leatherbacks utilize are prone to excessive erosion (J. Spotila 2004), increasing the cost of nesting conservation programs as imperiled nests are often relocated (Burkholder and Slagle 2015).

Leatherbacks feed on a variety of soft-bodied animals, including siphonophores, tunicates, and some crabs, though their primary prey is jellyfish (J. Spotila 2004). Unlike loggerheads, leatherbacks spend their entire lives devoted to feasting on jellyfish and do not experience dietary shifts between life-stages.

In the oceanic zone, leatherbacks take advantage of oceanographic features that aggregate their prey, such as mesoscale eddies, oceanic fronts, and other areas of

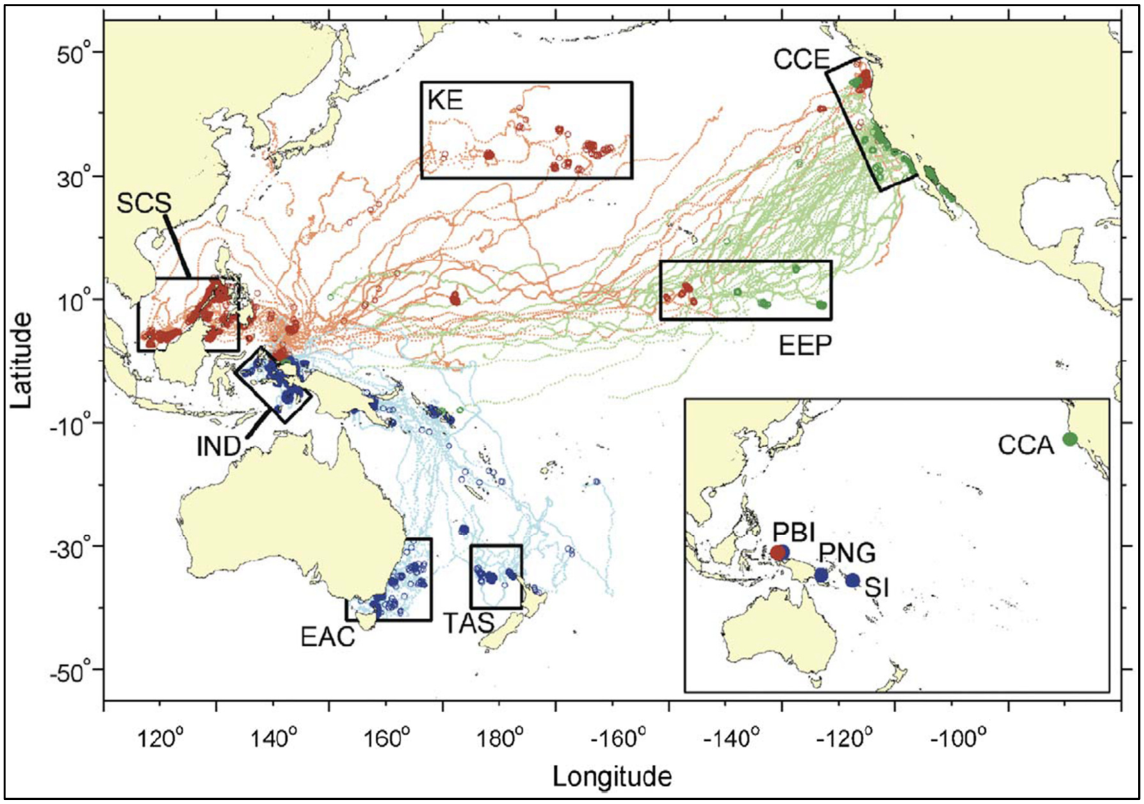

Figure 3 Tracks of tagged leatherback turtles from western Pacific nesting beaches and one eastern Pacific foraging ground. The map depicts inter-nesting and post-nesting movements, emphasizing the differences of high-use foraging areas for arboreal summer (red tracks) and winter (blue tracks) nesting populations, as well as turtles tagged in foraging grounds off central California (green tracks). Deployment locations shown in insert: Papua Barat, Indonesia (PBI), Papua New Guinea (PNG), Solomon Islands (SI), central California (CCA). Black boxes demarcate important ecoregions typically associated with oceanographic features (e.g., frontal features along boundary currents, convergence zones): South China, Sulu, and Sulawesi Seas (SCS), East Australia Current Extension (EAC), Tasman Front (TAS), Kuroshio Extension (KE), equatorial eastern Pacific (EEP), and California Current Ecosystem (CCE). (Adapted from Benson, E et al. 2011)

retention (Luschi, Sale, et al. 2003, Benson, Eguchi, et al. 2011). Females will spend the 2-3 year remigration interval traveling within and along these features, sometimes swimming thousands of kilometers between temperate and tropical foraging areas seasonally (Figure 3; Benson et al. 2011). Leatherbacks travel equal or greater distances to return to nesting beaches. Some individuals of the Indonesian nesting population have been tracked foraging as far away as the coast of California (Benson et al. 2011). Unfortunately, the oceanographic features that attract leatherbacks also attract commercially targeted species, such as tuna and swordfish. This, coupled with their long distance migrations, bring leatherbacks into the realm of different fisheries, leading to interactions with an array of gear types throughout their lifetimes.

Threats

Predation

Sea turtles have evolved various life history strategies to deal with natural levels of predation at land and on sea. As they pass through different life stages, sea turtles become increasingly more resilient to the threats of predation. By producing large quantities of eggs, their life histories accommodate high mortality rates in the first several years of life, relying on higher survival of later life stages. As eggs, hatchlings, and small juveniles, both species are especially susceptible to predation. On land, foxes, raccoons, coati mundis, ghost crabs, and other predators will target eggs and hatchlings (J. Spotila 2004). Once they enter the sea, hatchlings face a new group of predators, including sea birds, large fish, and sharks. Once turtles reach the juvenile stages, and continue to grow, fewer predators are able to prey upon them. As adults, only large sharks and saltwater crocodiles are capable of taking a turtle (J. Spotila 2004). Unfortunately, turtles no longer have to overcome only the natural threat of predation. Anthropogenic threats have become a much larger issue.

Overfishing/Harvesting

Overfishing has been, and continues to be, the primary force behind anthropogenically derived extinctions throughout the world's ocean ecosystems (Jackson, et al. 2001). Historically, turtle populations have been devastated by exploitation through overfishing and overharvesting of eggs, e.g., 90% decrease in nesting loggerheads in Japan (Peckham et al. 2007), and near or total extirpation of leatherback populations throughout the Pacific (Spotila, et al. 2000). Globally, progress has been made through international cooperation, such as the Convention of International Trade in Endangered Species of 1973 (CITES), as well as national programs, like Mexico's total ban on the harvest of turtles and eggs in 1990 (Aridjis 1990). Unfortunately, the illegal harvest of turtles and eggs persists, continuing to have negative impacts on sea turtle populations (Seminoff, Jones, et al. 2003, Wilson and Tisdell 2001).

While direct harvesting of sea turtles has been greatly reduced globally, indirect capture of turtles in other fisheries (i.e. bycatch) still poses a significant, and arguably the most serious, threat to sea turtles.

Bycatch

Though overfishing has historically been the primary driver of reductions of global sea turtle populations, the most consequential modern threat to sea turtles is unintentional capture, and often subsequent mortality, through fisheries bycatch (Lewison, et al. 2004, Wallace, Lewison, et al. 2010, Finkbeiner, et al. 2011, Lewison, et al. 2013). In the United States, minimum annual estimates of bycatch related sea turtle deaths were still 4600 after numerous bycatch mitigation measures (e.g., turtle excluder devices (TEDs), spatial and temporal closures) were introduced between 1996 and 2008. This estimate is down significantly (96%) from an estimated 71,000 deaths annually between 1990 and 1996 prior to the implementation of such measures (Finkbeiner, et al. 2011). Both large-scale, commercial, and small-scale, artisanal, fisheries can have significant rates of bycatch, resulting in a substantial number of interactions dependent on fishing effort.

Bycatch is not exclusive to any particular fishing technique or gear type and can result from the use of longlines, gillnets, seine nets, trawls, and even traps (Finkbeiner, et al. 2011, McClellan, et al. 2011). When Wallace et al. (2010) analyzed bycatch rates for three general categories of gear type, longlines displayed the highest impact, followed by trawls and gillnets, with longline rates more than doubling the other two combined (based on bycatch per unit effort). The impacts of each gear type varied regionally, and total bycatch for a specific region and gear type did not necessarily reflect the most severe interactions, i.e., the highest bycatch per unit effort (BPUE) did not necessarily align with the highest number of turtles taken per fishery (Wallace, et al. 2010). This highlights that even though bycatch rates for a particular large-scale fishery may be low relative to similar (or different) style fisheries, it can still have a large impact due to the sheer volume of the catch. Similarly, small-scale fisheries that land relatively small catches and yet have a high BPUE can have significant impacts on local turtle populations.

The severity of the impacts for each gear type differs by species and within species, which can be observed for each distinct population (Cheng and Chen 1997, Finkbeiner, et al. 2011, Lewison, et al. 2013). To characterize these affects, Wallace et al. (2010) split each species into Regional Management Units (RMUs) based on geographically explicit populations identified by tracking data and genetics. Of the two species investigated, loggerhead RMUs experience higher bycatch rates than leatherback RMUs (Lewison, et al. 2013). Globally, Lewison et al. (2013) found gillnets to be the primary gear type affecting leatherbacks, with loggerhead bycatch being more often associated with longlines.

Survival rates are important to consider when evaluating the impacts associated with specific fisheries. Purse seine fisheries, for example, provide an interesting case. Though the net may surround turtles, the open-air design of purse-seines allows for turtles (and other bycatch) to be easily released with minimal stress to the animal. Thus, even with a global operation and intensive fishing effort, purse-seine fisheries have low impacts on turtle populations (IATTC 2004). Conversely, some small-scale gillnet fisheries have bycatch mortality rates of up to 90% (Gass 2006). These operations, then, pose a much more serious threat to turtle populations, even if they are limited to small geographic regions.

Accurately addressing global bycatch, and even accurately estimating bycatch impacts and rates in specific fisheries, is constrained by the lack of standardized and widespread data (Davies, et al. 2009, Wallace, Lewison, et al. 2010, Lewison, et al. 2013). Fisheries observers play a vital role in the quantification and understanding of global bycatch, including turtles (Benaka and Dobrzynski, The National Marine Fisheries Service's National Bycatch Strategy 2004). However, their primary focus remains in the large commercial fleets of developed countries, resulting in massive oversight of bycatch rates in smaller fisheries and less developed countries (Lewison and Crowder 2007, Peckham, Diaz, et al. 2007). As such, most estimates are extrapolations from low sampling values and can vary widely.

Light-Pollution

Though sea turtles are only exposed to detrimental artificial lighting during the short time they spend on nesting beaches as hatchlings and nesting females, the impacts of these interactions on turtle populations can be profound. Witherington and Martin (2000) define the peculiar nature of light pollution, stating, “For sea turtles, artificial light is best described not as a toxic material but as misinformation.” This definition highlights the disorientation that artificial lights induce in hatchling sea turtles while attempting to locate the ocean (Witherington and Bjorndal 1991, B. E. Witherington 1992, Witherington and Martin 2000, Tuxbury and Salmon 2005, Donahou 2014).

Upon emerging from the sand, hatchlings use visual cues to orient themselves toward the brightest horizon and crawl immediately towards the sea (on naturally dark beaches) (Salmon and Witherington 1995, Lohmann, et al. 1997). However, with the introduction of an artificial light source, nesting females and hatchlings can misinterpret the visual cues coming from the sea and instead orient themselves away from the sea, which often leads to exhaustion, excessive depredation, and dehydration (Witherington and Martin 2000, Tuxbury and Salmon 2005, Lorne and Salmon 2007). Even if a disoriented hatchling eventually makes it to the water after a landward crawl, their ability to swim offshore is diminished (Lorne and Salmon 2007).

Light-pollution may also deter females from nesting on brightly lit beaches (B. E. Witherington 1992, Witherington and Martin 2000, Mazor, et al. 2013). Witherington (1992) indicated that not only light intensity influences nesting beach selection/deterrence, but also the type of light used (i.e. the spectrum of wavelengths emitted by each light source). Yellow low-pressure sodium-vapor (LSP) lamps, which emit long wavelength light, have a minimal effect on site selection by nesting females; while broad-spectrum lights, or those that emit an abundance of short wavelength and ultraviolet light, drastically reduce the amount of nests within the lighted area (B. E. Witherington 1992, Witherington and Martin 2000).

Habitat Destruction

Sea turtle habitat destruction is a multifaceted issue linked to varying life history phases of each species. As turtles navigate these different phases, (e.g., post-hatchlings and juveniles in pelagic convergence zones, recruiting later to neritic oyster beds and coral reefs as sub-adults, and then seeking appropriate nesting sites) they encounter different types of habitat degradation. In pelagic convergence zones, terrestrial and marine based anthropogenic debris gather alongside turtle prey and shelter, ranging in size from microplastics less than 1 cm in diameter (Cozar, et al. 2014) to huge commercial trawl nets and other pieces of derelict fishing gear (DFG; McElwee, Morishige and Donohue 2012). This debris can affect, often fatally, turtles through ingestion and entanglement (Carr 1987b, Derraik 2002).

Nearshore habitats can suffer degradation from DFG and plastic debris, as well as new dangers. On coral reefs, DFG cannot only ensnare turtles, but it can also destroy the habitat itself, by smothering and breaking corals (Donohue, et al. 2001, U.S. Environmental Protection Agency 2002). Seagrass beds can take up to four years to recover from watercraft propeller damage, while direct interactions between turtles and watercraft are usually far more serious and often fatal (Davenport and Davenport 2006).

The effects of coastal development, e.g., habitat loss and light pollution, are detrimental to nesting activities for turtles. On these beaches, obstructions (e.g., beach furniture) and disorientation from lighting deter females from nesting and interfere with hatchlings successfully making their way to the ocean (Taylor and Cozens 2010). On a more global scale, sea-level rise is reducing the total nesting area available (Fish, et al. 2005, Fish, et al. 2008). Coastal armoring further complicates this issue by removing the upper beach, preventing the inland retreat of beach structures (Dugan, et al. 2008, Fish, et al. 2008) and placing more obstacles onto the beach to deter nesting females (Witherington, Hirama and Mosier, Barriers to sea turtle nesting on Florida (United States) beaches: linear extent and changes following storms 2011).

Tourism

Coastal environments attract the largest annual percentage of tourists, creating a strong demand for coastal resorts, roads, and supporting infrastructure (Davenport and Davenport 2006). This infrastructure, along with millions of tourists, put high demand on coastal ecosystems through myriad impacts, including: pollution from sewage, antifouling compounds, and hydrocarbons (Davenport and Davenport 2006); light pollution (Lake 2008), large coastal structures, sand removal, and other nesting deterrents (Taylor and Cozens 2010); large influxes of plastics and other debris (Sheavly 2010); damage from watercraft, including propellers and anchors (Williams 1988); as well as many others. Poorly, or irresponsibly, planned resorts on nesting beaches can inflict long-lasting damage, both to the habitat, as well as to local culture and politics (I. Cheng 1995, Venizelos and Corbett 2005).

Aside from land-based tourism impacts, harassment of marine turtles and other organisms from divers or snorkelers (Meadows 2004) and other consumptive activities can have negative consequences for local ecosystems (Davenport and Davenport 2006).

Sea turtle sightings are a draw for tourists and divers at coral reefs and tropical destinations (Lucrezi, Saayman and van der Merwe 2013), yet divers often impose significant environmental impacts on local reefs. Coral reef damage from divers can result from direct contact, whether intentional or unintentional, by touching or handling with hands and kicking with fins, as well as by re-suspending sediment that may then settle on and smother corals (Barker and Roberts 2004). Areas of intense diving consistently show higher numbers of broken and damaged corals along with lower coral cover (Tratalos and Austin 2001, Hasler and Ott 2008). However, with increased awareness and education, divers may also serve a critical role in marine conservation. Divers value high biodiversity, including sea turtle presence, offering hope for an increased conservation push in threatened coral habitats (Schuhmann, et al. 2013).

Pollution

Anthropogenic influxes into the marine environment, from derelict fishing gear (U.S. Environmental Protection Agency 2002) to runoff of pesticides (Storelli and Marcotriciano 2000), continue to cause declines in sea turtle populations. Marine protected areas and fishing gear regulations can help mitigate some of these impacts, but these issues will largely need to be managed through targeted policies aimed at ocean cleanup and pollution prevention. Though pollution is a serious threat to all species of sea turtle, this topic will not be thoroughly discussed in this review.

Modeling

Population modeling has been used to estimate population trends in North Atlantic loggerheads, allowing for more informed management decisions. These models were based upon a population projection matrix, which is an adaption of the Leslie-Lewis matrix that is commonly used in mathematical ecology (Ricklefs and Miller 2000). The design of the matrix is adapted to reflect the lifecycle of the organism and is structured according to an age or life stage-based classification scheme, i.e., age-classified (A_1) or stage-classified (A_2) (Caswell 2001). In age-classified models, each step of the matrix is equal to one year, with the probability of surviving each year represented by P and the reproductive output of each year represented by F. Stage-classified models follow the assumption that individuals within a set of ages are subject to identical survival rates and reproductive values (Caswell 2001). They share the reproductive parameter, F, but differ in the handling of survival rates. For many organisms, more than a single year is spent in each stage. To account for the process of surviving but remaining in the same stage class, the stage-classified matrices use P as the probability of surviving and remaining within the same stage i, and G as the probability of moving onto the next stage $i + 1$.

$$A_1 = \begin{bmatrix} F_1 & F_2 & F_3 & F_4 & F_5 \\ P_1 & 0 & 0 & 0 & 0 \\ 0 & P_2 & 0 & 0 & 0 \\ 0 & 0 & P_3 & 0 & 0 \\ 0 & 0 & 0 & P_4 & P_5 \end{bmatrix}$$

$$A_2 = \begin{bmatrix} P_1 & F_2 & \mathrm{F}_3 & \mathrm{F}_4 & \mathrm{F}_5 \\ G_1 & P_2 & 0 & 0 & 0 \\ 0 & G_2 & P_3 & 0 & 0 \\ 0 & 0 & G_3 & P_4 & 0 \\ 0 & 0 & 0 & G_4 & P_5 \end{bmatrix}$$

Traditionally, loggerhead population models have been constructed using a stage-classified model (Crouse, Crowder and Caswell 1987, Crowder, et al. 1994, Caswell 2001). However, to account for the long delays spent in each stage, as well as to more accurately model the complex reproductive patterns exhibited by sea turtles (i.e., the variation in years between nesting), Heppell (1998) reconfigured the matrices into an age-classified scheme. This has since become the population modeling approach that is employed by the Turtle Expert Working Group (TEWG) (National Marine Fisheries Southeast Fisheries Science Center 2001).

When considering models and population growth rates, it is important to recall that they are not an exact representation of the population, but a guide. They do not take into account density-dependent or other compounding factors and thus should not be taken as an absolute. That is, if the population growth rate is positive, it will not continue to grow exponentially into perpetuity. Similarly, negative growth rates do not necessarily dictate the imminent demise of the population.

Statement of Significance

Of the large volume of studies that discuss sea turtle conservation strategies, the majority of them limit their focus to the application of a single method to a single species, a single method on a few species, or multiple methods on a single species. There is a paucity of literature that investigates the application of multiple conservation techniques on multiple populations of different marine turtle species. Through this paper, I will provide a comprehensive review of the efficacy of two classes of conservation strategies (nesting beach conservation and fisheries management) for two turtle species (loggerhead and leatherback turtles) throughout their respective ranges and life stages. These two species were chosen due to their contrasting life history traits, shared conservation risks, and their wide distribution.

Additionally, the most current loggerhead population models were last published in 2003 (Heppell et al.). To evaluate the effectiveness of conservation strategies since, these models were updated in this paper with parameters from the current literature.

Methods

A thorough search was performed using Web of Science and local library resources to find peer-reviewed publications, technical reports, conference proceedings, and resources relevant to each of the selected conservation categories: nesting beach conservation and fisheries management. The results of a subset of these studies were tabulated for review (Appendices I-IV). Study inclusion was determined from credibility of results (e.g., bycatch mitigation studies that did not encounter turtles were not included). These varied results were compared to determine best practices within each category, which were delineated by the threats that they address. Nesting beach conservation was split between mitigating the effects of light pollution, habitat degradation, and depredation. Due to a lack of quantifiable results, the first two were not tabulated. Fisheries management was delineated by fisheries type (i.e., longlines, passive nets, and trawl nets).

To translate the effectiveness of conservation strategies into their effects on real populations, a case study was employed using a model population of North Atlantic loggerheads (Richardson and Richardson 1979). Keeping with the structure established

Table 1 Stage durations used in updated models. [1]Heppell et al. 2003; [2]Ramirez et al. 2015; [3]Avens et al. 2015; [4]Assumed to compensate for age at sexual maturity.

	Oceanic Immature	Small Neritic Immature	Large Neritic Immature	Age at Sexual Maturity
Model 1	10[1]	11[1]	13[1]	35[1]
Model 2	12[2]	11[1]	12[4]	36[3]
Model 3	12[2]	11[1]	12[4]	36[3]
Model 4	12[2]	11[1]	12[4]	36[3]

by the TEWG, I employed an age-classified model around the lifecycle framework used in Heppell, Crowder, et al. (2003) (Figure 4).

Having defined the model structure, I evaluated four different model parametrizations. Model 1 parameterization utilized a set of parameters from Heppell, Crowder, et al. (2003) that most closely resembled the currently understood ASM estimated by Avens et al. (2015). Stage duration parameterization for the three new models (Models 2-4) followed updated parameters from the literature (Table 1). If there did not exist an updated duration value, the missing parameters were assumed and fitted to sum to ASM. Survival rate parameterization similarly utilized updated values from the literature. There was not an updated survival rate for large neritic juveniles; thus, the survival rate from Model 1 was used for Models 2-4 for this age class (Table 2).

The resulting matrices are age-classified, with P_i values, the probability of surviving to the next year, along the subdiagonal (Table 3). In initial breeding year, which occurs in the column corresponding to the ASM (Table 1), all females are assumed to nest with the surviving proportion progressing to the next year.

I modified the calculation of each element in the matrix to compensate for the remigration intervals exhibited by the breeder classes. Each year, the females observed on

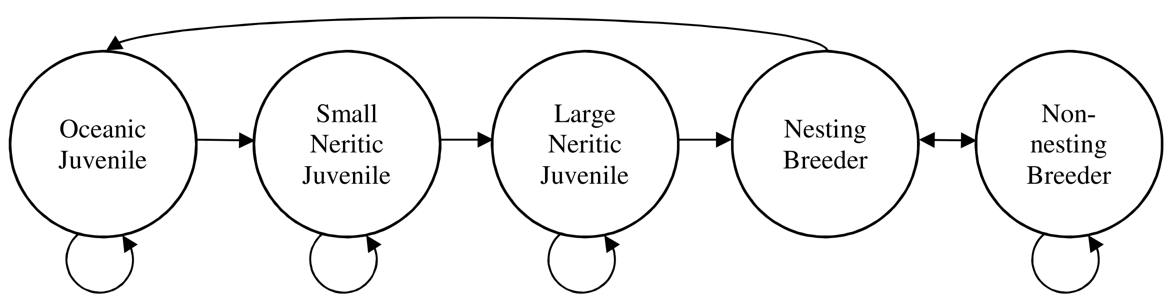

Figure 4 Life cycle graph of loggerhead matrix models. Adapted from Heppell, Crowder et al. 2003.

Table 2 Annual survival rates used in updated models. [1]Heppell et al. 2003; [2]Bjorndal, Bolten and Martins 2003; [3]Sasso et al. 2006; [4]Monk, Berkson and Rivalan 2011; [5]Sasso, Epperly and Johnson 2011.

	Oceanic Immature	Small Neritic Immature	Large Neritic Immature	Nesting Breeder	Non-nesting Breeder
Model 1	0.875[1]	0.7[1]	0.8[1]	0.85[1]	0.85[1]
Model 2	0.72[2]	0.81[3]	0.8[1]	0.85[4]	0.85[4]
Model 3	0.72[2]	0.81[3]	0.8[1]	0.41[5]	0.41[5]
Model 4	0.72[2]	0.81[3]	0.8[1]	0.6[5]	0.6[5]

the beach constitute 44% of the total nesting female population, due to variations in remigration rates, i.e., 3%, 56%, 31%, 7%, and 3% of nesting females remigrate after one, two, three, four, and five years, respectively. I incorporated the transition probability (ψ_i, eq. 1) into the calculation of each element, including reproductive output (F), where r is the remigration rate for that age (A_3) (Monk, Berkson and Rivalan 2011).

$$A_3 = \begin{bmatrix} 0 & 0 & 0 & F_4 \times \psi_i & F_5 \times \psi_i \\ P_1 & 0 & 0 & 0 & 0 \\ 0 & P_2 & 0 & 0 & 0 \\ 0 & 0 & P_3 & 0 & 0 \\ 0 & 0 & 0 & P_4 \times \psi_i & P_5 \times \psi_i \end{bmatrix}$$

For juvenile classes, ψ_i is equal to 1. The probability of surviving to the next year is P.

$$\psi_i = \frac{r_i}{1 - \sum_{j=1}^{i-1} r_j} \quad (1)$$

The transition probability, ψ_i, was incorporated into all parameters for the last five columns of the breeder class. This created a remigration cycle within the last five columns, with the appropriate proportion of remigrating breeder females nesting and returning to the beginning of the cycle each year, and the rest proceeding to the next year of the remigration cycle (Table 3).

Within the fecundity term, F (eq. 2), the annual survival rate for the first year of life (0.6747), sex ratio (0.5), and reproductive output are accounted for. Reproductive output is equal to the average number of nests per female (4.1) multiplied by the average eggs per clutch (115) (Turtle Expert Working Group 1998). The annual survival rate is included in the calculation of fecundity because the model is built using a pre-breeding census. That is, the census of the population occurs immediately before each breeding

Table 3 Truncated age-classified matrix using survivorship values from Heppell, Crowder, et al. 2003. The matrix was necessarily expanded and modified to reflect suggested variations in these variables when running other models. Each survivorship parameter is repeated per the stage duration. For example, if the initial class duration were twelve years, 0.745 would be repeated for twelve columns along the subdiagonal. The top row represents fecundity. The final six columns represent the breeder class, with the nesting breeders accounted for along the other horizontal row. The final five columns represent the remigration cycle, accounting for the variation in remigration rates exhibited by loggerheads. The non-nesting breeders are represented along the subdiagonal starting in the fifth to the last column. The parameters for each of the elements in the remigration cycle incorporate ψ, the transition probability for that remigration year.

Oceanic Juv.		Sm. Neritic Juv.		Lg. Neritic Juv.		Age at Sexual Maturity					
1	…	13	…	24	…	36	Remigration Cycle				
0	**…**	**0**	**…**	**0**	**…**	**159.06**	**4.77**	**91.83**	**120.26**	**111.34**	**159.06**
0.875	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
⋮	**⋱**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
0	**0**	**0.81**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
⋮	**0**	**0**	**⋱**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
0	**0**	**0**	**0**	**0.8**	**0**	**0**	**0**	**0**	**0**	**0**	**0**
⋮	**0**	**0**	**0**	**0**	**⋱**	**0**	**0**	**0**	**0**	**0**	**0**
0	**0**	**0**	**0**	**0**	**0**	**0.85**	**0.026**	**0.491**	**0.643**	**0.595**	**0.850**
0	**0**	**0**	**0**	**0**	**0**	**0**	**0.825**	**0**	**0**	**0**	**0**
0	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.359**	**0**	**0**	**0**
0	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.2073**	**0**	**0**
0	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0**	**0.255**	**0**

cycle, thus the eggs and hatchling class has already been through one year of life prior to the census (Heppell, pers. comm.).

$$F = nests \times eggs \times survival\,rate \times sex\,ratio \qquad (2)$$

To evaluate each projection matrix, the annual population growth rate (λ) was determined as the dominant eigenvalue of each matrix. Both λ and the stable stage distribution (*w*, the right eigenvector) were calculated using a custom Python script (Python Software Foundation. Python Language Reference, version 3.4). To evaluate the effect changes in each parameter (i.e., survival rate and stage duration) have on λ, an elasticity analysis was performed. This begins with calculating the sensitivity matrix, which, derived by using the left and right eigenvectors, *w* and *v* (eq. 3). Here, $\langle w, v\rangle$ is the scalar product of the eigenvectors.

$$Sen_{kl} = \frac{\partial\lambda}{\partial a_{kl}} = \frac{(v_k, w_l)}{\langle w, v \rangle} \quad (3)$$

Because the orders of magnitude differ greatly between parameters (e.g., F = 159.061 and P_1 = 0.875) the sensitivities for these values are difficult to compare. To determine the relative contribution of the parameters, these values were scaled into elasticities, where all the elasticities of the matrix elements sum to 1 (Velez-Espino, Fox and McLaughlin 2006).

$$\varepsilon_{kl} = \frac{a_{kl}}{\lambda}\frac{\partial\lambda}{\partial a_{kl}} \quad (4)$$

The elasticity of each stage is thus the sum of elasticities for each element in that stage.

$$\varepsilon_{stage\ i} = \sum_{j=T_{initial}}^{T_{final}} \varepsilon_j \quad (5)$$

Sensitivity and elasticity calculations were performed using MATLAB (R2012a, ver. 7.14.0; The Mathworks, Inc. 2012). Construction of the matrices, projection, and elasticity calculations followed the methods of Caswell (2001). The new values used for survivorship and stage duration were attained by searching the published literature for updated parameters (Tables 1 and 2).

Review

Nesting Beach Conservation

The protection and monitoring of nesting beaches is imperative to population dynamics of turtles as it directly affects the beginning of the lifecycle (Appendix I). Without effective beach conservation programs, nesting success can be impaired due to avoidance by nesting females and increased hatchling mortality (Mann 1977, Taylor and Cozens 2010, Roe, et al. 2014). Risks to nesting beaches arise from a variety of sources both naturally and anthropologically derived. Among natural sources, severe storms, depredation, natural erosion and accretion, and tides can pose severe, though typically unavoidable, threats to sea turtle nests (Boulon, Jr. 1999, B. E. Witherington 1999). Anthropogenic threats, such as light-pollution, vehicular traffic, and beach grooming, can often be addressed (B. E. Witherington 1999, Witherington and Martin 2000); while others, such as beach nourishment (Rumbold, Davis and Perretta 2001) and coastal armoring (B. E. Witherington 1999, Witherington, Hirama and Mosier 2011), may have lasting detrimental effects.

The construction of sea walls, jetties, rock revetments, and other forms of coastal armoring may impact sea turtle nesting success (Witherington, Hirama and Mosier 2011). Jetties and groins can influence longshore flow, disturbing natural beach accretion cycles, and may lead to downstream erosion of coastal features (Mohanty, et al. 2012, Pietrafesa 2012). This erosion can reduce or eliminate important nesting beaches. Similarly, artificial nourishment of nesting beaches with mined sand can alter the grain structure of the beach, making it less suitable for nesting. Poor placement of sea walls can reduce the availability of suitable beach above the high-tide line (HTL), forcing turtles to lay clutches closer to the HTL (Witherington, Hirama and Mosier 2011b) in areas where they can be exposed via erosion or inundated with seawater, leading to high mortality or total loss of the clutch.

Though coastal armoring is typically permanent, its effects can be mitigated through the modification or removal of groins, dikes, berms, etc. (Cereghino, et al. 2012). However, the most effective way to address coastal armoring is through the strict enforcement of conservative setback requirements (i.e., the minimum distance a structure must be setback from a specific shoreline feature), which prevents the construction of permanent structures adjacent to the beach or on primary dunes. This can eliminate or significantly reduce the need for coastal armoring (B. E. Witherington 1999). The Coastal Zone Management Act of 1972 (CZMA) issued a national policy concerning the protection and preservation of the coastal zone, including the protection of coastal and dune systems, while allowing and encouraging the states to develop and implement the most appropriate management strategies for their respective regions with the provision of federal funds (16th U.S. Congress 2005). As of 2012, fourteen of the twenty-three coastal and Gulf coast states, excluding Alaska, had instituted statewide coastal setback requirements based on a set distance in feet from various coastal features (e.g., vegetation lines or high tide lines) or long-term annual erosion rates (Randall and deBoer 2012).

In Broward County, Florida, construction of coastal armoring is not permitted during marine turtle nesting season (Broward County Rules and Procedures for Coastal Construction). There is a 50-foot setback line in the county, and local legislation acknowledges the necessity of the natural beach and dune systems. However, construction of coastal armoring is still permitted if certain parameters are met: namely, that there is not a significant impact to the system and that the structure is at direct risk from natural coastal processes (Florida Department of Environmental Protection 2012).

Lighting

Mitigation of light pollution offers some of the easiest solutions, but many are difficult to enforce, as many coastal light sources are privately or commercially owned (Lake 2008). The most definitive way to combat light pollution is to simply turn coastal lighting off. However, since significant portions of coastal lighting are designed for safety, this is often not feasible. Instead, modifications to lighting schemes can often limit the impacts of artificial lighting on the beach or eliminate the spillover of light into coastal beach systems altogether (Witherington and Martin 2000). As both hatchling sea turtles and nesting females are impacted by bright, broad-spectrum lighting, switching to low-pressure sodium-vapor lamps can reduce the impact of coastal lighting, especially if the light-source is shielded and directed away from the beach.

High mounted coastal roadway lighting can be especially problematic, illuminating long stretches of beaches when the lighting is poorly shielded, especially in the absence of natural dune systems. To counteract this, lighting can be embedded into the roadway, preventing the scattering of light onto adjacent beach systems (Bertolotti and Salmon 2005). When dune systems are present, shielded, low-mounted lighting can also be acceptable (Witherington and Martin 2000).

Dune restoration is critical to aid in the shielding of artificial light sources. In natural systems, hatchling turtles are capable of using dune silhouettes to aid in seafinding as the silhouettes reinforce the contrast in brightness between land and the seaward horizon. However, if artificial lighting cannot be eliminated, but can be reduced in brightness, the silhouettes from high dune systems may still provide adequate cues to properly orient hatchlings (Tuxbury and Salmon 2005). When reestablishing dunes, native coastal vegetation is necessary to ensure long-term establishment and retention of the dune system. The use of ornamental plants in substitution of native vegetation is discouraged due to the risk of spreading invasive species (Awale and Phillott 2014). Until the vegetation has matured to allow for adequate light shielding, light screens (e.g., shade cloth, privacy fences) may be used to enhance dune silhouettes (Witherington and Martin 2000). In areas where artificial lighting cannot be quickly mitigated, shielded pathways that orient hatchlings seaward from the nest can be used as a temporary solution (Witherington and Martin 2000).

Detrimental artificial lighting at private residences largely originates from patio lighting, interior lighting (visible through beach-facing windows) and general area lighting (e.g., used to illuminate pool areas). Commercial sources of lighting can be more intense, originating from sources that directly illuminate the beach intentionally, along with unexpected sources intended to light restaurants, bars, stairwells, walkways, or parking lots, along with other areas. These latter sources are common at beachfront hotels and resorts, where lighting considerations were limited to the benefits and safety of guests, and not to the potential impact to sea turtles (Knowles 2007, Lake 2008).

The implementation of lighting ordinances can be an effective way to reduce lighting impacts from beachfront homes and commercial properties (Figure 5). Broward County, FL implemented sea turtle lighting ordinances in 2000. Donahou (2014) observed an average annual decrease in hatchling disorientation from 2006-2011 for the county, with

Figure 5 Before and after photographs of two commercial properties that incorporated sea turtle friendly lighting (STFL). (Adapted from Barshel, et al. 2014.)

stronger improvements seen on beaches that are more compliant. Donahou also noted the existence of disorientation "hotspots" that correlated to coastal areas where compliance was minimal. Similar improvements were seen in Sarasota County and Manatee County. Barshel et al. (2014) reported hatchling disorientation at multiple coastal commercial properties and found that disorientation events dropped from an annual range of 50-300 to zero after the implementation of turtle safe lighting.

Depredation

When depredation by mammals or other large predators is the primary threat to a nest, there exists a variety of proposed solutions. Among the most apparently straightforward management techniques for nesting beaches is the direct manipulation of sea turtle nests. This can be accomplished in a variety of ways (e.g., cages, translocation) and for a variety of reasons (e.g., excessive depredation risk, light pollution). However, the best treatment of turtle nests is no treatment (in situ). Relocating eggs leads increases egg mortality due to embryonic detachment to the egg wall, and non-natural egg chambers can affect sex ratios, among other complicating factors (Boulon, Jr. 1999). Thus, assuming natural conditions are intact and depredation risks are low, manipulation will likely not improve the hatching success of an *in situ* clutch. Unfortunately, these conditions rarely exist on sea turtle nesting beaches, and thus existing threats need be mitigated to achieve minimum hatchling mortality. To combat depredation, some management strategies include aversive conditioning, predator removal or control, or construction of cages around nest sites (Boulon, Jr. 1999). The latter two options are common, while the former has had limited applications.

Aversive conditioning by conditioned taste aversion involves using treated bait (e.g., chicken eggs inoculated with toxic chemicals to condition turtle egg predators) to teach predators to avoid targeted prey items. Conditioned taste aversion (CTA) has seen mixed success using various techniques, thus its effectiveness is controversial. In controlled studies, mongoose (Nicolaus and Nellis 1987) and foxes (Baker, et al. 2007) were found to develop CTA to chemical-laced baits, but this was not long lasting in the mongoose. However, when CTA was compared against other protection methods against raccoons, there was no significant reduction in nest depredation rates (Appendix II; Ratnaswamy, et al. 1997).

Hatchling sea turtles fall prey to a number of predators on nesting beaches, including raccoons *Procyon lotor,* dogs *Canis lupus* spp., feral hogs *Sus scrofa*., coati mundis *Nasua* sp., foxes *Vulpes vulpes*, etc. (Boulon, Jr. 1999). When choosing a predator control method, care must be taken to ensure a thorough understanding of complex food web connectivity. Total predator removal may not be necessary, as often only a small percentage of individuals within a predator population will specialize in preying on turtle nests. Targeting and removing these problem individuals can be highly effective in reducing nest loss. With raccoons, nest predation is considered a learned behavior, and thus only the individuals within the population who have been taught to seek nests need to be removed. In Ten Thousand Islands, Florida, it took the removal of only 16 raccoons to reduce nest depredation from 76-100% in 1991-1994 to 0% in 1995 and 1996 (Appendix II; Garmestani and Percival 2005). However, at Canaveral National Seashore, Florida, 50% of the resident raccoon population was removed without any reduction in nest depredation (Ratnaswamy, et al. 1997).

Removal of top predators may have unintended consequences throughout an ecosystem, e.g., detrimental increases in herbivore abundance resulting in overgrazing away from the coastal system (Letnic, Ritchie and Dickman 2012); inadvertent increases in secondary predator abundance, resulting in increased depredation on turtle nests. The latter scenario was demonstrated by Barton and Roth (2008) in Florida where low raccoon abundance was correlated with higher ghost crab densities, which occurred where the nest depredation rates were highest.

To control predator species that are not native to the area (e.g., feral hogs) or whose abundance can be reduced without significant ecological impact, possible options are removal/eradication through shooting or trapping and relocation/euthanasia (Boulon, Jr. 1999, Garmestani and Percival 2005, Engeman, Duffiney, et al. 2010). Shooting strategies can include public hunts, but shooting should only be used in unpopulated areas, while keeping in mind a possible response from animal rights organizations. Similarly, trapping using toxic bait or embarking on poisoning campaigns may unintentionally kill non-target species, including other locally important species, domesticated animals, and children if such strategies are not carefully controlled (Boulon, Jr. 1999).

Feral hogs are not native to any nesting beach, and thus their removal can be pursued aggressively. In the U.S., NMFS promotes the complete elimination of feral hog populations on sea turtle nesting beaches as part of the recovery plan for the northwest Atlantic loggerhead population (National Marine Fisheries Service and U.S. Fish and Wildlife Service 2008). On Cayo Costa Island, Florida, baiting and trapping/shooting effectively reduced nest depredation from 74% to 15% after two years of removal efforts (Appendix II; Engeman, et al. 2010).

The spatial and temporal strategies of predator removal programs should be routinely refined to ensure an effective use of resources, both human and financial. This should be accomplished through continual monitoring to ascertain when, where, and which removal efforts should be applied in order to achieve maximum effects with minimal labor, leading to optimal hatching success. Prior to predator removal, nest depredation on Hobe Sound National Wildlife Refuge, Florida was at 95%. Initial removal of predatory raccoons and armadillos reduced depredation to 42%. Predator removal tactics were then optimized using passive tracking techniques, further reducing depredation to 28% (Appendix II; Engeman, et al. 2003).

To prevent digging of the nest from the surface, cages constructed of metal mesh laid over the nest work well. Cage design can be tailored to fit specific protection needs, but typically follows a general pattern. In a basic design, metal mesh can be laid over the nest and secured at the corners with stakes or buried 5-10 cm below sand surface, effectively preventing digging entry from above (Ratnaswamy, et al. 1997, Yerli, et al. 1997). The simplicity of this approach makes it preferable in areas with limited resources or without exceptionally persistent predators. On Canaveral National Seashore, Florida, screening of 2/3 of nests caused a 20-50% reduction in nest depredation compared to predator removal and CTA, which were both ineffective (Appendix II; Ratnaswamy, et al. 1997). Another comparison study on Dalyan Beach, Turkey, found that nest screening resulted in 0% fox depredation versus 63% depredation of unscreened nests (Yerli, et al. 1997).

In some areas, medium-sized predators (e.g., raccoons) may still be able to access nests, especially shallow nests, by digging between the mesh or entering from the side (Addison 1997). In these scenarios, a rectangular cage of galvanized metal or plastic mesh buried 15-30 cm into the sand, with the bottom 15 cm bent outwards, will create a more substantial barrier to digging and tunneling predators (Figure 6; Addison 1997, Boulon, Jr. 1999). Addison and Henricy (1994) tested screens versus cages on Key Island, Florida. Of the screened nests, raccoons were still able to partially depredate 11.4% and fully depredate 13.6% of experimental nests, while depredation of caged nests was 0% and 3.6%, respectively. A similar study by Kurz et al. (2011) found that plastic mesh screens were 25% less effective than cages at preventing nest depredation for highly motivated foxes on Bald Head Island, North Carolina. However, under normal conditions, both methods resulted in 0% depredation versus 33% for untreated nests (Appendix II).

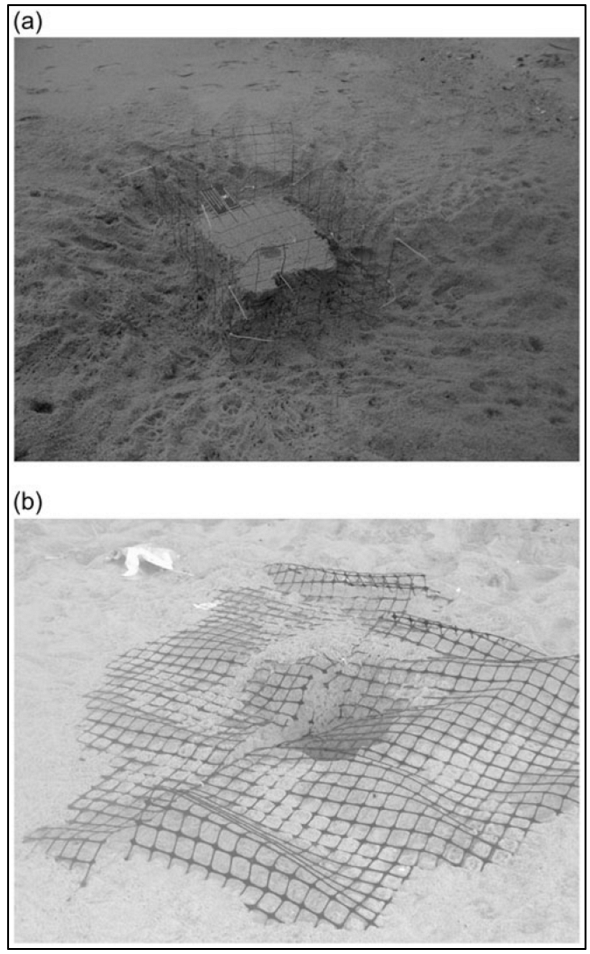

Figure 6 Comparison of cages (a) versus screens (b) in protecting nests against depredation. (a) The fox(es) attempted to dig into the egg chamber from all sides, but were prevented due to buried cage design. (b) Fox(es) were able to access the egg chamber by digging through, and eventually breaking, the mesh screen covering the nest. (Adapted from Kurz, Straley and DeGregorio 2012.)

When choosing the mesh for cages or screens, predator size should dictate mesh size, e.g., for medium-sized mammals such as dogs and raccoons, 5x10 cm mesh is suitable. Smaller mammals, such as mongoose, will require smaller mesh; however, mesh of this smaller size must be removed prior to hatching to ensure hatchlings are not prevented from reaching the sand surface (Boulon, Jr. 1999). The larger mesh size does not appear to impede hatchling emergence (McElroy 2006). Metal cages have been shown to distort the magnetic field within caged nests, but the effects of this distortion on turtles as hatchling or later in life have yet to be investigated (Irwin, Horner and Lohmann 2004).

Over the years, many nesting beach conservation programs have experimented with nest relocation, and when, where, and how often it should be applied. Through this experimentation, multiple issues have been discovered, such as polarization of sex ratios due to egg chamber temperature (J. Spotila 2004), and increasing the threat of damage from storms by aggregating nests on a small section of beach. Additionally, hatchling success from relocated nests is typically lower than *in situ* nests due to the stress of transport and reburial. In a review by Grand and Beissinger (1997), they found that hatching success for loggerheads was greater for in situ nests than relocated nest for a wide range of international nesting beaches, specifically in the absence of depredation risks. Similarly, hatchling success in Broward County, Florida was 83.6% for in situ nests and 69.4% for nest relocated due to erosion or inundation risks (Burkholder and Slagle 2015).

Thus, relocation of nests should always be a last resort and should only be undertaken if leaving the nest in its natural location will lead to imminent and near-total mortality of the clutch. Appropriate instances for nest relocation are nests laid 10 ft or less from high tide line or near areas of known high natural erosion; nests located near artificially lighted areas where lighting impacts cannot be mitigated, especially near highways; nests in areas undergoing active beach nourishment or sand mining; or areas where threats from depredation or poaching are too great and cannot be easily dissuaded (Boulon, Jr. 1999, Burney and Ouellette 2005). When necessary, however, nest relocation can be an effective management tool. In Gandoca Beach, Costa Rica, poaching was reduced from 100% to 15.5% annually with the use of nest relocation, combined with camouflaging of nests and nightly monitoring (Chacon-Chaverri and Eckert 2007).

To ensure effective application, relocation methods should adhere to strict protocols. Improper egg transport during translocation may lead to detachment of the embryo from the egg case, resulting in embryo death. To prevent this, eggs should be collected within twelve hours of being laid; ideally, the eggs should be collected as nesting is occurring. Once collected, eggs should be reburied within six hours. If it is necessary to transport the eggs large distances, they should be secured in a sturdy container (e.g., a bucket) and insulated from vibration and hard shocks. Care should be taken to maintain consistent egg orientation as well, i.e. the same part of the egg should always face up. Artificial nests should mimic as closely as possible the shape, depth, and egg deposit order as the original nest. When choosing reburial sites or hatchery locations, it is important to consider ground temperature and moisture content due to the likelihood of sexual polarization of the clutch due to nest chamber temperature. For a detailed description of collection and reburial procedures, see Boulon, Jr. (1999) and the Florida Fish and Wildlife Conservation Commission Marine Turtle Guidelines (2007).

After nests are successfully relocated, additional nest protection like placing a cage or wire mesh, especially in a hatchery location where concentrated nests create a tempting source for depredation are commonly used. In hatcheries that are regularly monitored, placing fine fabric mesh can prevent the infiltration of insects, including sarcophagus flies (Chacon-Chaverri and Eckert 2007).

Hatchling release needs to be facilitated if nests are moved to hatcheries without a clear, unencumbered path to the sea, or if the hatchery is far removed from the original nesting beach. Hatchery personnel should monitor nests every 30-60 minutes during the expected emergence period to allow for release as soon as possible after emergence. To avoid marine and terrestrial predators from being able to predict release areas, consecutive release sites should not be within several hundred meters of each other. Turtles should be allowed to crawl across the beach and enter the water unaided to facilitate natal beach imprinting. When transporting hatchlings prior to release, or if immediate release cannot be accomplished, turtles should be kept in a dark, damp, cool, cloth sack to discourage crawling and prevent the expenditure of energy reserves (Boulon, Jr. 1999).

Fisheries Management

Historically, a leading cause of sea turtle population decline worldwide is overharvesting of turtles, both at sea and on beaches, as well as the harvesting of turtle eggs (Jackson, et al. 2001, Lewison and Crowder 2007). While the direct taking of sea turtles has been drastically reduced, thanks largely to CITES and CMS, fisheries continue to have devastating effects on turtle populations through incidental bycatch (Bourjea, et al. 2008, Finkbeiner, et al. 2011). Fortunately, progress has been made. In the US, fisheries-specific mitigation measures resulted in an ~94% decrease in fishery-related turtle deaths between 1990 and 2007 (Finkbeiner, et al. 2011).

Aside from negative environmental impacts, bycatch also inflicts economic consequences by generating additional costs, impacts fishers by tarnishing their public image, causing conflicts within the fishing community, and can lead to smaller and lower quality yields (Hall, Alverson and Metuzals 2000). Therefore, it is important to include stakeholders (i.e., fishers), as well as resource managers, in discussions concerning fisheries management approaches, as they are the ones who must actually follow and enforce regulations (deReynier, Levin and Shoji 2010).

Unfortunately, solutions to bycatch are not universal due to the variations in the global fisheries landscape. Fishery styles vary immensely, from small artisanal set nets, to massive pelagic purse seines, to pelagic long lines that are kilometers long and have up to several thousand hooks. Fisheries vary seasonally and spatially, fluctuate in technological advancement and limitations of access to physical and financial resources required for retrofits. There are also fundamental differences in strategies between ocean basins even within the same style of fishery (e.g., pelagic longline fisheries). Similarly, each species interacts with different types of fisheries and gear throughout their lifetime.

The Food and Agriculture Organization of the United Nations (FAO) has delineated fishing gear into eleven major categories. The three of these that primarily impact sea turtles are “hooks and lines”, which includes longlines; “gillnets and entangling nets”, from which I will discuss drift nets and pound nets; and “trawl nets” (Figure 7; http://www.fao.org/fishery/topic/1617/en).

Hooks and Lines

In the Atlantic Ocean, pelagic longline fisheries (PLF) focus their efforts around major submarine (e.g., shelf breaks) and oceanic features (e.g., edges of fronts and warm-core rings) that are often concentrated in a geographically small area (Boggs 2003). In contrast, PLF in the Pacific Ocean are not nearly as limited. They typically operate in waters deep enough (>4000m) that submarine features do not play a role in habitat formation for target species and instead operate along large ocean frontal boundaries (e.g., North Pacific Transition Zone, Subtropical Frontal Zone) (Boggs 2003).

Thus, the magnitude and complexity of the issue posed by global bycatch complicates the universal application of mitigation strategies and policies. Additionally, when policies are agreed upon and enacted in a particular fishery, there is often push back from the fishery due to the short-term economic constraints posed by the policy, such as investing in and implementing new technologies (e.g., TEDs, circle hooks) (Hall, Alverson and Metuzals 2000).

The most common sources of bycatch arise from trawl and longline fisheries, where exposure to gear typically occurs in the neritic and pelagic zones, respectively. Loggerhead bycatch rates are generally higher than those of leatherbacks (e.g., 90% higher in a study done with pelagic longlines by Santos et al. (2013)). However, with significantly lower population sizes for leatherbacks, especially in the Pacific, these lower catch rates still pose a serious risk (Roe, et al. 2014).

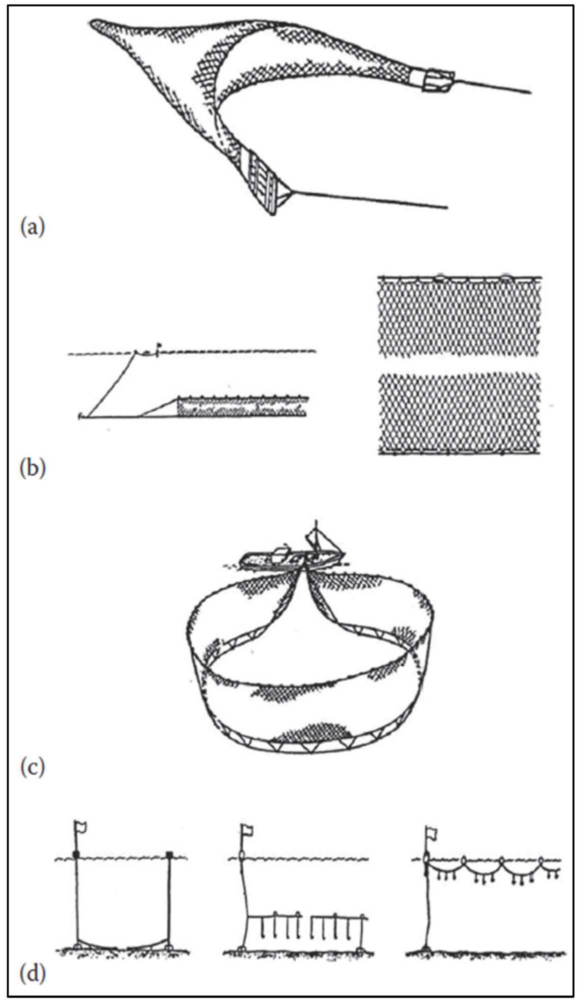

Figure 7 Illustrations of four general types of fishing gear known to impact sea turtles: (a) trawl nets, (b) passive nets, (c) purse seines, (d) longlines. (Adapted from Lewison, et al. 2013.)

Gear modifications are among the most common measures implemented to mitigate bycatch. These differ between fisheries, such as hook choice, in longline fisheries, and Turtle Excluder Devices (TEDs), which have shown tremendous success in trawl fisheries (Epperly 2003). For longlining, several mitigation methods have been proposed, including changes to hook type, bait type, and set depth.

There is a wide variety of hooks utilized by PLF, which vary in general shape (J versus circle) and can be offset to various degrees (Figure 8). The traditional and most commonly used hook type is the J hook, which has a high incidence of bycatch for

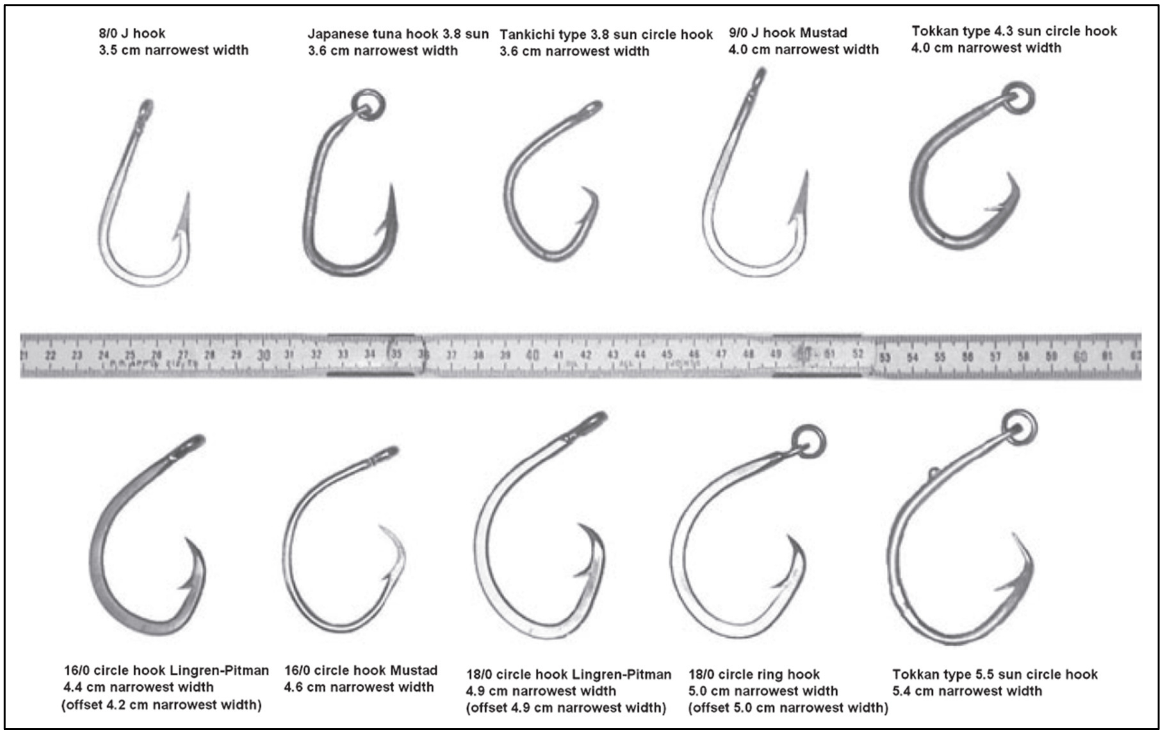

Figure 8 A sample of hook variety utilized in longlining operations. Many of these were used in referenced studies. (Adapted from Gilman, Zollett, et al. 2006.)

loggerhead turtles via hooking in the mouth or throat, and leatherbacks via hooking externally on the flippers (Santos, et al. 2013). A common alternative that has shown great promise for bycatch reduction is circle hooks. In the Brazilian PLF, Sales et al. (2010) observed a 55% and 65% reduction in bycatch rates for loggerheads and leatherbacks, respectively, when changing from 9/0 J-hooks to 18/0 circle hooks. Similarly, 16/0 circle hooks caught 70% less juvenile loggerheads than similarly sized J-hooks in the Mediterranean swordfish longline fishery (SLF) (Appendix III; Piovano, Swimmer and Giacoma 2009).

An additional possible benefit of using circle hooks over J-hooks is the effect this can have on hooking location. Smaller J-hooks are more easily swallowed, often leading to deep-hooking, i.e., the hook being set in the lower esophagus or stomach. Larger circle hooks, on the other hand, are more difficult to swallow and are more likely to hook in areas of the mouth (e.g., tongue and jaw) or on external structures (e.g., flippers) (Parga 2012, Parga, et al. 2015). In the Azores SLF, for example, rates of deep-hooking were 13% for circle hooks, as compared to 60% and 52% for J-hooks and Japanese tuna hooks, respectively (Appendix III; Bolten and Bjorndal 2005).

When turtles are deep-hooked, it can be substantially more difficult to remove the hook and line, and post-release mortality is assumed to be greater. Casale, Freggi, and Rocco (2008) evaluated the effects of longline hooking location and attached branch line length for rescued turtles collected in the central Mediterranean. They found that hooks lodged in the lower esophagus/stomach had a more severe impact than those hooked in the upper esophagus or mouth, and that hooks associated with attached branch lines 51.5 cm or greater had a lower chance of survival.

Indeed, hooking in the mouth and mandible is often preferred, as these hooks are easier to remove than swallowed hooks, possibly promoting post-release survival (Read 2007, Alessandro and Antonello 2010). However, Parga (2012) debates the issue and suggests that damage to sensitive structures within the mouth, including the jaw joint and the glottis, could lead to potentially fatal disease and infection. She proposes that by having strong, muscular, and resilient esophagi, sea turtles may be more likely to survive deep-hooking, so long as the hook does not become lodged close to the heart or near major blood vessels (Parga 2012). In this regard, there exists a discussion as to the least consequential hooking location.

Swimmer et al. (2013) used satellite tracking to estimate post-release mortality based on hook location for loggerheads caught in the North Pacific Ocean PLF. They found no significant difference in post-release days at liberty for turtles that were deep- versus shallow-hooked (Swimmer, et al. 2013). Though days at liberty may not accurately represent mortality, these results support the notion that deep-hooking is not inevitably more fat al than shallow-hooking. These implications are more relevant to loggerheads, as leatherback interactions with longline gear primarily result in entanglement or external hooking (Watson, et al. 2004).

In conjunction with changing hook-type, variations in bait type have been found to influence sea turtle bycatch rates and hooking location. In the majority of studies surveyed, the traditional bait in PLF is squid, with mackerel as the experimental alternative. The application of 18/0 circle hooks with mackerel bait in the Northeast Distant SLF resulted in a 90% reduction in loggerhead bycatch and a significant shift in hooking location to the mouth when compared to the J-hook and squid control. For leatherbacks, this combination resulted in a 65% reduction compared to the control hook and bait, whereas J-hooks with mackerel had a 66% reduction. Incidentally, seven of the eight leatherbacks that were hooked in the mouth, rather than externally, were captured on circle hooks (Watson, et al. 2005). In the Hawaiian SLF, this same combination reduced bycatch by 90% and 83% for loggerheads and leatherbacks, respectively, with a similar change in hooking location for loggerheads (Gilman, et al. 2007). Santos et al. (2013) had comparable results in the Portuguese SLF in the South Atlantic, where circle hooks baited with mackerel reduced bycatch by 87.5% for loggerheads and 100% for leatherbacks. In this study, circle hooks again shifted hooking location to the mouth for loggerheads, while leatherbacks were consistently hooked in the flipper or entangled in the line (Appendix III).

It is believed that the shift in hooking location that is correlated to bait type arises from the methods that turtles (especially loggerheads) apply when eating different prey. Stokes et al. (2011) found that captive-reared loggerhead turtles would tear fish bait, usually stripping it from the hook, where squid was usually consumed whole resulting in higher rates of hook interaction for squid bait.

Unfortunately, transitioning to circle hooks is not effective in preventing sea turtle bycatch in all fisheries. In the Uruguayan pelagic longline fishery, Domingo et al. (2012) found that the use of circle hooks had no significant impact on bycatch of either loggerheads or leatherbacks. In 2001 and 2002, Bolten and Bjorndal (2003) evaluated the performance of straight and offset J-hooks, along with 16/0 and 18/0 circle hooks in the Azores SLF. They found no significant difference in bycatch rates by hook type, though more loggerheads were deep-hooked on J-hooks than circle hooks. Additionally, a study using captive reared loggerheads showed a significant increase in ingestion rate for hooks less than 51mm wide, regardless of hook shape. This width corresponds to 16/0 circle hooks and 11/0 J-hooks. (Watson, Hataway and Bergmann 2003).

Furthermore, though reduction of sea turtle bycatch may occur, utilizing circle hooks in some fisheries can significantly reduce the catch rate of target species and is therefore not a viable management strategy (Sales, et al. 2010, Amorim, et al. 2014). Thus, careful evaluation of hook choice impacts should be done in all fisheries before regulations are implemented to assure effective management and economic viability (Gilman, Zollett, et al. 2006).

Legislation involving circle hooks has been realized in several fisheries. As of July 2004 in the U.S. Atlantic PLF, the NMFS implemented regulations that mandated the use of 16/0, or larger non-offset circle hooks in all regions, except for the Northeast Distant waters, where 18/0 or larger circle hooks with less than a 10°offset were mandated (Stokes, Epperly and McCarthy 2012). In addition, these regulations required vessels to possess and utilize equipment designed to handle and release hooked or entangled sea turtles. While this is an important step in conservation legislation, the U.S. tuna fisheries only constitute 3% of the global tuna production (Gilman and Lundin 2008). Fortunately, the IATTC and several of its partner organizations, including the Overseas Fisheries Cooperation Foundation of Japan and the World Wildlife Foundation (WWF), have sought to improve and self-regulate the interactions that the tuna longline fisheries in the eastern Pacific have with sea turtles. This project, Resolution C-04-07 of the IATTC (Inter-American Tropical Tuna Commission 2004), began in 2004 and involved monitoring turtle bycatch rates, evaluating mitigation strategies, and educating the industry through informational material and educational meetings. In 2008, the World Wildlife Fund began managing the project. Since the start of the project, approximately 700 longline vessels from nine countries in the Eastern Pacific PLF have adopted the use of circle hooks and hook removal equipment voluntarily (World Wildlife Foundation 2015).

Water temperature has been shown to influence bycatch rates as well, due to the temperature preferences of turtles (Secretariat of the Pacific Community 2001, Boggs 2003, Watson, Epperly and Shah, et al. 2005, Gilman, Zollett, et al. 2006). Several mapping tools and programs make use of this temperature preference by tracking sea surface temperature (SST) and other oceanographic features, and creating maps for turtle “hot spots”, or areas with increased risk for interactions between sea turtles and fishing gear (Howell, et al. 2008, Roe, et al. 2014, Howell, et al. 2015).

In 2006, NOAA created a tool (TurtleWatch) for the Hawaii-based PLF that uses operational longline fishery characteristics, bycatch information, satellite-tracking data for loggerheads, and remotely sensed SST to create maps that indicate areas of high bycatch potential (Howell, et al. 2008). In 2013, satellite-tracking studies on leatherbacks were incorporated into the studies, allowing the modeling to be effective for both turtle species in the North Pacific (Howell, et al. 2015). This tool maps the SST zones where species- specific interaction chances are high (17°-18.5°C for leatherbacks and loggerheads, 22.4°-23.4°C for leatherbacks) (Howell, et al. 2015). Roe et al. (2014) used similar modeling techniques to predict bycatch hotspots throughout the entire Pacific Ocean. Cambie et al. (2012) used a GIS-based method to develop maps for the Southern Italian Coast. The data from these studies suggest that it is difficult to avoid turtle bycatch altogether because both species, and especially leatherbacks, forage along and follow the same transitional frontal systems as the target species, (e.g., swordfish in the North Pacific Subtropical Frontal Zone) (Howell, et al. 2015). Thus, fishers are unlikely to avoid the areas of high interaction risk, as these are also the areas where target catch per unit effort (CPUE) is highest (Howell, et al. 2015).

TurtleWatch is available in three languages to fishers and managers of the Hawaii-based SLF. While the initial fishing ground selection may not avoid the recommended zones, fishers are more likely to adapt their plan using the product once the hard cap limits for sea turtle interactions (annual catch of 17 loggerheads or 16 leatherbacks for the Hawaiian SLF) are being approached (Howell, et al. 2015). Reaching these limits, introduced in 2004 by NMFS, warrants the institution of section 7 of the ESA and closure of the shallow set fishery for the rest of the calendar year (NMFS 2004), which occurred in 2002-2004 and again in 2006 (Gilman, et al. 2007). As these limits are applied to the entire fleet and not per vessel, fleet communication programs are important tools to disseminate information quickly within fisheries pertaining to incidental bycatch of protected species. Gilman, Dalzell, and Martin (2006) reviewed the bycatch data for the U.S. North Atlantic LSF and found a 50% reduction in BPUE for J-hooks after the industry began a fleet communication program that included reporting turtle/longline gear interactions. This suggests that voluntary fleet communication programs could substantially reduce bycatch while providing economic incentive (i.e. avoiding fishery closures). The latter of these is vital to encourage participation in the programs and to ensure compliance with mandated regulations (O'Keefe, Cadrin and Stokesbury 2013).

Set depth, the depth at which fishing gear is deployed, is another factor in rate of sea turtle bycatch in longline fisheries. Set depth is usually fishery specific, however, SLF typically employs shallower sets (<40m) than tuna fisheries (>100m), due to preferred target catch habitat (Secretariat of the Pacific Community 2001, Boggs 2003, Gilman, Zollett, et al. 2006). Thus, regulation of set depth is only relevant in fisheries that meet certain criteria (e.g., where target catch can still be maintained using deeper sets). Otherwise, other mitigation strategies must be explored.

Gillnets and Entangling Nets

Passive net fisheries (e.g., gillnets, trammel nets), both commercial and small-scale, can have substantial impacts on sea turtle populations (Gass 2006, Peckham, Maldonado-Diaz and Koch, et al. 2008, Gilman, Gearhart, et al. 2010, Murray 2013, Peckham, Diaz, et al. 2007). Moreover, impacts from small-scale and artisanal fisheries may be comparable or greater than those of commercial fleets (Peckham, et al. 2007, Peckham, et al. 2008, Gilman, et al. 2010). Small-scale fisheries employ over 99% of the world's fishers, 95% of which are in developing countries where conservation policies are often weak, if present, with little resources for regulation and enforcement (Berkes 2001). This makes observing and quantifying the impacts of these fisheries difficult.

Loggerhead turtles more commonly interact with passive net fisheries on their neritic foraging grounds. Pacific loggerheads that nest in Japanese rookeries spend considerable time foraging in the waters of Baja California Sur, Mexico (Peckham, et al. 2007). In this area, Peckham et al. (2007) estimated a minimum annual bycatch of 1000 loggerheads per year in just two small fishing fleets - a rate that rivals ocean wide commercial fishing operations.

Due to their largely pelagic life history, leatherbacks primarily interact with passive net gear off nesting beaches during breeding season. This is especially prevalent in the Western Atlantic, which hosts the largest leatherback nesting population (Spotila 2004, Gearhart and Eckert 2007, Gearhart, Eckert and Bergmann 2009). In Trinidad's artisanal gillnet fishery, mortality from the over 3000 gillnet interactions was estimated at 27-34% in 2000 and 32% in 2005 (Gass 2006, Lum 2006). This level of interaction between gillnets and leatherbacks inflicts financial hardship on local fishers due to the economic loss associated with net repair costs and lost fishing time (Gass 2006).

Various strategies have been proposed to combat bycatch associated with these high mortality fisheries. Two studies in the Trinidad surface drift net fisheries evaluated the effects of manipulating the set depth of drift nets (Gearhart and Eckert 2007) and shortening the height of the drift nets (Gearhart, Eckert and Bergmann 2009). Only the latter was successful in reducing leatherback bycatch rates (by 11-74%), though with a notable reduction in target catch (35-55%). However, the significant reduction in turtle interactions made the experimental height adjustment more attractive to local fishers (Gearhart, Eckert and Bergmann 2009). Peckham et al. (2009) also investigated changes to net profile, as well as tie-down length, and similarly saw no significant change in sea turtle bycatch rates.

Other drift net strategies investigated included elimination of buoys on net float lines (Peckham, Maldonado-Diaz and Lucero, et al. 2009), changing the marking lights from white to red (Gearhart, Eckert and Bergmann 2009), and incidental take permits and fishery closures (Byrd, Hohn and Godfrey 2011) (Appendix IV). Of these, only the latter showed significant reductions in turtle bycatch rates, though fishery closures certainly reduced target catch volume. Eradication of their use is also an option, as seen in Morocco, which banned the use of driftnets in 2012 in recognition of their indiscriminate and destructive impacts (Appendix IV; Benhardouze, Aksissou and Tiwari 2012).

Pound nets can also impact nearshore marine turtle populations. When the top of the "pound" or trap portion of the net is open to the air, there is less of an issue, as the turtle can simply wait to be released (assuming they do not become entangled in the framing nets). However, when the net design incorporates an underwater bag, the long soak times typically result in mortality for trapped turtles. Soak times for this style net are usually very long due to the complex nature of their design.

Variations in design to mitigate turtle interactions and mortality include modifications to leader height and integration of release doors on the top of submerged nets. Leader modifications can allow the turtle to swim over the leader without being redirected to the pound and can reduce the risk of the turtle becoming entangled in the leader itself. In the Virginia pound net fishery, one study reduced the leader height by 2/3, resulting in only one turtle interaction, as opposed to 21 interactions with the unmodified leader, most of which were loggerheads. The one interaction with the shorter leader was the only leatherback in the study, which became tangled in a rope attached to the leader (Figure 9; Silva, Dealteris and Milliken 2011). In Japan, Abe and Shiode (2009) performed trials with captive turtles to develop set-releasing doors for underwater bags. By utilizing pointed-top nets that directed trapped turtles toward a self-releasing door, approximately 80% of turtles were able to escape (Appendix IV; Abe and Shiode 2009).

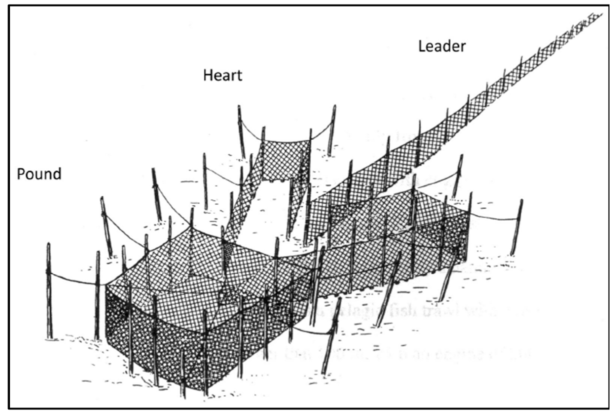

Figure 9 Illustration of a pound net. In the studies reviewed, the height of the leader and pound sections of the net were reduced. (Adapted from Silva, Dealteris and Milliken 2011.)

Though interactions with purse-seines can result in mortality, the overall bycatch risk for sea turtles from purse-seine fisheries is low. If turtles are encircled when the net is set, the lack of a top allows turtles to breathe until they can be released from the net. The IATTC estimated the sea turtle mortality associated with large purse-seine operations in the Eastern Pacific Ocean to range from 46-172 annually for 1993-2002, with only one leatherback in the ten-year period and two loggerheads annually (IATTC 2004, FAO Fisheries and Aquaculture Department 2009).

Trawl Nets

Trawl fisheries have long been recognized as having a major impact on sea turtle populations, with shrimp trawls far exceeding the impacts of any other fishery in the U.S. by up to two orders of magnitude, prior to regulation. In the Southeastern Atlantic, U.S. Atlantic, and Gulf of Mexico (GOM) shrimp trawl fisheries, bycatch interactions and mortality events for all turtle species were estimated at 340,500 and 69,300, respectively, prior to regulation (Finkbeiner, et al. 2011). Due to soak times averaging 2-3 hours at a time, turtles entrained within a trawl net suffer high mortality rates (Jenkins 2012). Fortunately, TEDs have proven highly effective at mitigating turtle bycatch in trawl fisheries (Lewison, et al. 2013). Unfortunately, even after the implementation of TEDs, bycatch interactions in trawl fisheries still far exceed those in other fisheries by two to three orders of magnitude (133,400) and mortality events are still one to two orders of magnitude higher (3700) (Finkbeiner, et al. 2011). Even after TEDs had been required for more than 5 years, 70-80% of dead turtle strandings on U.S. beaches were related to shrimp trawl fisheries in 1995 (Crowder, Hopkins-Murphy and Royle 1995).

Turtle excluder devices have proven to be effective in trawl fisheries throughout the world, including Australia (Brewer, et al. 2006), Kuwait (Al-Baz and Chen 2014), the Mediterranean (Alessandro and Antonello 2010), and the United States (Jenkins 2012). In other regions, trawl fisheries still capture thousands of turtles annually (e.g., greens and hawksbills in the Red Sea (Teclemariam, et al. 2009)), while others have seen major improvements with the implementation of TEDs (e.g., green, olive ridleys, and flatbacks in the Northern Prawn Fishery in Australia (Brewer, et al. 2006, Barwick 2011)). Unfortunately, though TEDs have been around for over 35 years and are used in over 40 countries, there is little published data concerning their use and effectiveness outside of the U.S. and Australia. Since loggerhead and leatherback turtles most commonly interact with trawl gear in the Northwest Atlantic, Gulf of Mexico (GOM), and Mediterranean, this section will largely focus on the data produced from U.S. research.

For the North Atlantic and the GOM, the National Marine Fisheries Service (NMFS) has largely been responsible for TED development and testing (Jenkins 2012). The NMFS began testing TED designs in 1976, leading to the release of the original NMFS TED in 1981 (Figure 10, Jenkins 2012). Since then, numerous modifications have been made to the initial design, and other variations developed for specific applications have been produced. For example, the Super Shooter TED, which was designed for shrimp fisheries, improved upon the NMFS TED by removing the external frame. Moreover, both the Modified Flounder TED and the Whelk TED II included openings on the bottom to prevent the exclusion of target catch (Figure 10; Jenkins 2012).
These last two designs were developed after the NMFS began testing TEDs developed by the shrimp fishery industry in 1987. Soon after, the NMFS focused its efforts on testing and took over the design aspect of TEDs. Thus, the NMFS changed their focus to testing, modifying, and approving industry designed TEDs rather than developing their own designs (Jenkins 2012). Through extensive collaboration, an array of commercially and privately developed TEDs and TED accessories became available that provided solutions to various fishery-specific issues, including TEDs designed to prevent clogging by vegetation (the Anthony Weedless TED) and accessories to prevent chaffing/tearing of the net along the bottom edge of the TED (the Darien Roller) (Figure 10; Jenkins 2012).

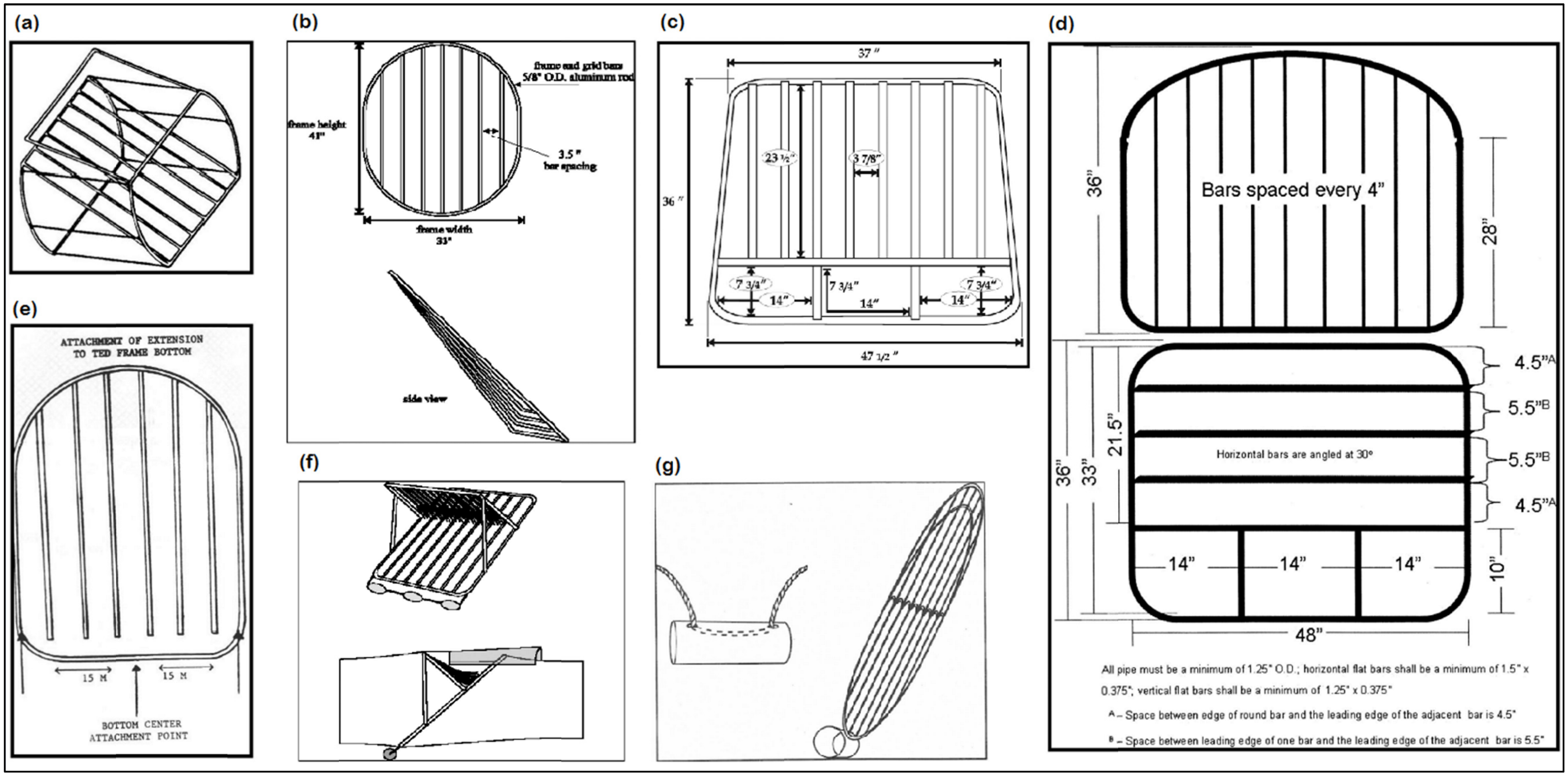

Figure 10 Illustration of various TED designs and accessories. (a) Original NMFS TED, (b) Super Shooter TED, (c) Whelk TED II, (d) Modified Flounder TED, (e) Anthony Weedless TED, (f) Pierce Shrimp Broom, (g) Darien Roller. (Adapted from Jenkins 2012.)

The NMFS required the use of TEDs in 1987 in shrimp trawl fisheries and expanded the regulations in 1992 to include fisheries in certain areas of Virginia and North Carolina when summer flounder (*Paralichthys dentatus*) season and turtle season overlap (NOAA 1992, Epperly 2003). The TED designs approved by NMFS prior to 2000 were 97% or more successful in the exclusion of small turtles, which were considered the cohort that had the highest interaction rate with trawl gear.

However, though TEDs had proven successful in experimental trials, there appeared to be up to a 50% disparity between the realized bycatch reduction (estimated from stranding data) and the target reduction rate in the U.S. (Lewison, Crowder and Shaver 2003). Discrepancies in exclusion rates began to be noticed in the initial years following TED regulations. In South Carolina, Hopkins-Murphy and Murphy (1994) noticed the percent of total strandings comprised of adult females increased from 12.8% to 18.9% in the two years following TED implementation in 1988, suggesting that not all classes of turtle were excluded uniformly. A later study by Epperly and Teas (2002) analyzed stranding data from the Sea Turtle Stranding and Salvage Network (STSSN) and found that 33-47% of loggerhead strandings were individuals who were too large to fit through the standard TED openings. Specifically, the carapace height of larger turtles exceeded the TED heights of 10 inches in the GOM and 12 inches in the Atlantic (Epperly and Teas 2002). To ameliorate this issue, TED size requirements were increased to accommodate larger turtles. Regulations requiring the use of these larger TEDs were implemented in 2003, resulting in considerable reductions in both bycatch and mortality associated with trawl fisheries for both loggerheads and leatherbacks (Table 4; Finkbeiner, et al. 2011).

In the Mediterranean, bottom trawl fisheries annually capture an estimated 30,000 and kill an estimated 8000 loggerhead turtles (Casale 2011, Sala, Lucchetti and Affronte 2011). Leatherbacks are uncommon in the region, with only 170 reported individual catches in the Mediterranean basin between 1981 and 2000, only 4.7% of which were caught in trawls (Casale, et al. 2003). Yet the promotion of TEDs remains difficult and regulations mandating their use in the Mediterranean are still non-existent (Domenech, et al. 2015). This is principally due to the large impact on target catch. Instead of the smaller shrimp species that are sought in U.S. waters, many of the fisheries in the

Table 4 Impacts of 2003 NMFS regulation mandating the use of enlarged TEDs in the U.S. trawl fisheries. These TEDs are capable of excluding adult loggerhead and leatherback turtles. Data take from Finkbeiner et al. 2011, Supplementary Material.

Fishery (Bycatch)	Loggerhead Pre-regulation	Loggerhead Post-regulation	Leatherback Pre-regulation	Leatherback Post-regulation
Mid-Atlantic Bottom Trawl	637.8	616	0	1
Mid-Atlantic Scallop Dredge	306	90	0	0
Mid-Atlantic Scallop Trawl	132	132	0	0
SE/Gulf of Mexico Shrimp Trawl	163160	23336	3090	520

Fishery (Mortality)	Loggerhead Pre-regulation	Loggerhead Post-regulation	Leatherback Pre-regulation	Leatherback Post-regulation
Mid-Atlantic Bottom Trawl	111	265	0	0
Mid-Atlantic Scallop Dredge	579	67.5	0	0
Mid-Atlantic Scallop Trawl	0	0	0	0
SE/Gulf of Mexico Shrimp Trawl	62294	647	2311	15

Mediterranean target finfish (Casale 2011). This becomes problematic, as TEDs developed for American fisheries will often exclude these target species (Lucchetti, et al. 2008). Efforts are being made to address these shortcomings and several TED designs have been evaluated for use in different Mediterranean bottom trawl fisheries. Lucchetti et al. (2008) evaluated five TED designs, only one of which did not substantially reduce target catch and showed additional value through improving the quality of the catch by excluding large debris. In the Adriatic Sea, another study evaluated four TED variations: adjustable, flexible, semi-rigid, and the rigid aluminum Supershooter TED (Sala, Lucchetti and Affronte 2011). Of the four, only the semi-rigid and Supershooter TED reduced discards of non-target species and debris without significantly reducing target catch. However, their effectiveness for turtle exclusion was inconclusive, as only one loggerhead was encountered during the study, though it was successfully excluded by the Supershooter TED being tested at the time (Sala, Lucchetti and Affronte 2011).

A major issue that has continuously plagued the success of TEDs is that many are disabled by fishers once at sea (Caillouet Jr., et al. 1996). It is difficult to regulate and monitor the large commercial fleets of the GOM and Western Atlantic. Due to this, fishers are able to sew shut the escape flaps on TEDs, rendering them completely ineffective, without detection by management agencies. Much of the motivation behind this meddling is the belief that TEDs lead to major losses of target catch.

In an attempt to change these opinions, proponents of TEDs often rebrand these devices as Trawling Efficiency Devices or simply classify them in the category of Bycatch Reduction Devices (BRD). The NMFS has employed the former since 1986 (Watson, Mitchell and Shah 1986) while internationally, the broader BRD is most often applied (Burke, Barwick and Jarrett 2012, Al-Baz and Chen 2014). The use of these terms highlights the additional benefits of TEDs, aside from their turtle exclusion function. TEDs have been shown to reduce other forms of bycatch, including elasmobranchs, large marine sponges, and non-targeted finfish. In contrast to many claims, several styles of TED have been found to increase target catch (Jenkins 2012). Continued testing and promotion of these benefits will serve to bolster the acceptance and effective use of TEDs in global trawl fisheries.

Modeling

There have been many changes to population management for sea turtles in the past few decades. To ensure that management strategies keep pace with population statuses, it is important to update the models these decisions are based on. For the Western North Atlantic loggerheads, there have been several updates since Frazer (1983) initially created a life table for this population (Crouse, Crowder and Caswell 1987, Crowder, Hopkins-Murphy and Royle 1995, National Marine Fisheries Southeast Fisheries Science Center 2001, Heppell, Crowder, et al. 2003). However, there has not been an update to these models since Heppell, Crowder, et al. revisited them in 2003. Since then, there have been numerous updates to annual survival rates (Table 2), as well

Table 5 Values of λ for each model with subsequent 30% reduction in mortality for specified life stages.

	Original Survival Rates	Small Neritic Immature	Large and Small Neritic Immature	All Neritic Juveniles and Adults
Model 1	0.953	0.984	1.007	1.014
Model 2	0.990	1.008	1.030	1.036
Model 3	0.962	0.982	1.006	1.011
Model 4	0.969	0.989	1.012	1.018

as more definitive estimates of age at sexual maturity (ASM) and stage durations for this population (Table 1).

For each model, the population growth rates, λ (derived from the eigenvalues and eigenvectors of the respective matrices), indicate that with current survival rate estimates, this population is still in decline ($\lambda \leq 1$; Table 5, Figure 11). In Model 2, with an adult annual survival rate 0.85, reducing mortality of the small neritic class results in a population growth of less than 1% annually. However, with each of the lower adult survival rates proposed by Sasso, Epperly and Johnson (2011), mortality must be reduced by at least 30% for both neritic juvenile classes (Figure 11). Reducing mortality of all neritic and adult age classes results in population growth of 1-3% annually (Table 5).

An evaluation of the elasticity analysis shows that the elasticity of λ to changes in annual survival rates for all juvenile stages is directly proportional to stage length (Table 6, Figure 12). That is, λ is more sensitive to survival rates for stages with longer durations (e.g., oceanic immature and large neritic immature in Models 2-4). The new models also

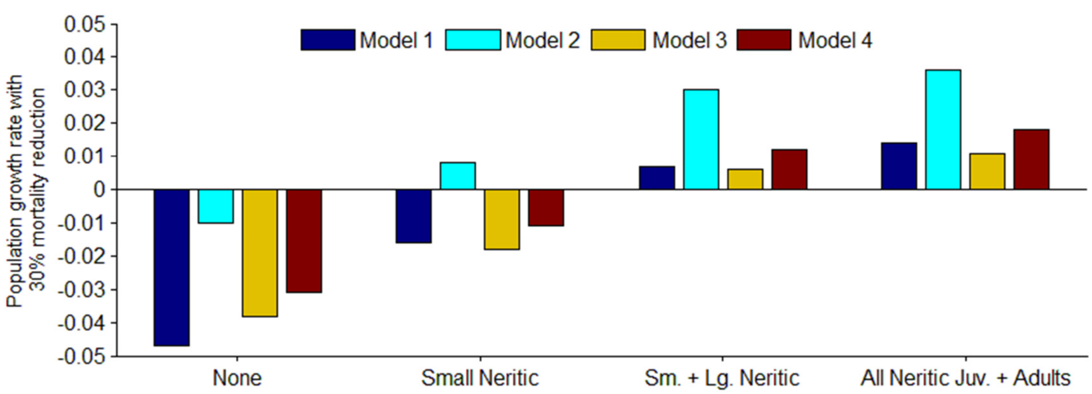

Figure 11 Population growth rate for each model with a 30% reduction in mortality for the indicated stages: no change; reduction in small neritic juvenile mortality; reduction in small and large juvenile neritic mortality; reduction in small and large neritic juvenile, as well as nesting and non-nesting breeders. Growth rate = λ – 1.

Table 6 Elasticity values of each stage for each model. Elasticity indicates the proportional impact each parameter has on λ.

	Fertility	Oceanic Immature	Small Neritic Immature	Large Neritic Immature	Nesting Breeder	Non-nesting Breeder
Model 1	0.0235	0.2355	0.2590	0.3061	0.0720	0.1038
Model 2	0.0242	0.2905	0.2663	0.2905	0.0531	0.0755
Model 3	0.0275	0.3301	0.3026	0.3301	0.0045	0.0051
Model 4	0.0270	0.3238	0.2968	0.3238	0.0127	0.0160

reinforce the conclusions that changes in egg and hatchling survival will have little impact on population trends (Crowder, Crouse, et al. 1994, Heppell, Crowder, et al. 2003). Similarly, with the low survival rates of adults in Models 3 and 4, changes to those rates are less impactful than improving survivorship in the juvenile age classes.

Discussion

Of the threats investigated here, the majority have been targeted by assorted management strategies in place across the globe. Each conservation method affords varying levels of protection during different life stages of marine turtles. Similarly, some methods are more effective as conservation tools for each species.

Nesting beach conservation exclusively targets threats posed against nesting mothers, developing eggs, and hatchling turtles. In general, these benefits extend to both species, while some threats are species specific. On beaches where measures are put into place to mitigate lighting, counteract poaching, or prevent damage associated with active

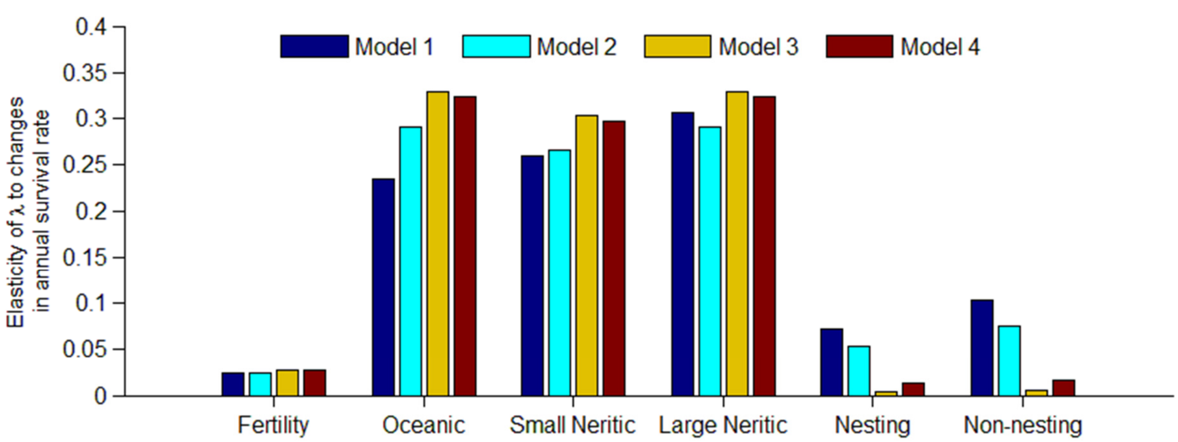

Figure 12 Elasticity values for the annual survival rates of each stage.

beach cleaning/nourishment, the benefits of these programs are shared by both species. Likewise, regulations that preserve and repair the structural integrity of nesting habitat, (e.g., limiting beach development, protection of dune systems, preventing or removing groins and jetties) benefit both species equally.

Conversely, loggerheads are typically the main benefactor from programs whose focus is on preventing depredation, though leatherback nesting may also occur on the same beach (e.g., South Florida). On nesting beaches where opportunistically oophagous (i.e., animals that supplement their diet with eggs when available) predators exist, loggerhead nests suffer much higher predation rates than leatherback nests due to the relatively shallow nest depths of the former. Leatherback nests are not completely exempt from depredation, however, especially on beaches with large (e.g., wild pig) or particularly ambitious diggers (e.g., large raccoons). On these beaches, predator removal programs can lead to exceptional increases in nesting success for both species.

The effectiveness of these programs varies, with caging programs having the largest influence on preventing nest depredation from foxes, raccoons, and armadillos. Nearly all of the studies showed significant reductions in depredation, with some completely preventing depredation. Predator removal programs, whether by trapping or shooting, are also largely successful, though success hinges on whether the correct individuals are targeted. For example, nest depredation has been identified as a learned behavior in raccoons. Thus, properly identifying and removing the animals that exhibit this trait results in much lower rates of depredation. Conditioned taste aversion has still not been shown to be an effective tool in reducing nest depredation.

In areas where poaching is still uncontrolled, beach monitoring and relocation programs serve both species. The protection from monitoring programs can extend to nesting females as well. Beach monitoring programs are important tools to track population trends and identify potential concerns. However, to become effective conservation tools, they must incorporate strategies that address these concerns (e.g., marking off nests to prevent disturbance from tourists or beach cleaning equipment, relocating nests laid in risky areas).

While protecting nesting habitat is crucial to supporting turtle populations, population modeling suggests that it may only be beneficial to a certain extent (Figure 12;

Crowder, Crouse, et al. 1994, Heppell, Crowder, et al. 2003). This is also supported by leatherback nesting trends in the Mexican Pacific, where intense nesting beach conservation over 20 years has not been able to reverse a severe and continuing reduction in leatherback nesting (Martinez, et al. 2007). However, it is difficult to discern whether a lack of response is due to factors affecting other life stages (e.g., bycatch in PLFs) or to a delayed age of sexual maturity.

Based on the loggerhead population models, it is difficult to make any inferences about what effects targeted conservation at particular life stages will have on population rates of leatherbacks. The stark contrast in life histories between the two species prevents the juxtaposition of models from one to the other. Notably, the rapid growth to sexual maturity for leatherbacks (13-29 yrs; Avens and Snover 2013) geatly alters the life stage duration parameters. Though the loggerhead models provide little insight for leatherback population trends, case studies (such as the one from the Mexican Pacific above) highlight that without conservation efforts which encompass multiple life stages of each species, population recovery is often infeasible.

The majority of the threats marine turtles face occur at sea by way of interactions with fishing gear. Longline, passive net, and trawl fisheries place significant burdens on turtle populations. However, due to the disparity in data reporting, with most observer data generated from PLFs, it is difficult to specify which fishery has the highest bycatch volume.

For loggerheads, there is some variation among which life stages are impacted by each major fishery type, with small and large neritic juveniles more often affected by trawl fisheries (Lewison, Crowder and Shaver 2003). Pelagic longline fisheries most commonly interact with loggerheads during their small neritic juvenile stages (Watson, Epperly and Shah, et al. 2005). Due to their fully pelagic lifestyle, leatherbacks interact with similar gear types throughout their lifetime. Unfortunately, bycatch reports do not typically include size measurements for individual turtles, complicating the characterization of stage-specific impacts (Finkbeiner, et al. 2011).

Bycatch rates in PLF are extremely low (e.g., .001-.034 turtles per 1000 hooks in the Pacific; Secretariat of the Pacific Community 2001), but due to the sheer number of hooks in the water, total bycatch numbers can still be devastating. The main management strategy in these fisheries is gear modification, i.e., changing from J-hooks to circle hooks. The goal with this modification is to prevent turtles from biting/ingesting the hooks or, at the very least, change hooking location from internal to external locations. Circle hooks were recommended by many authors for bycatch mitigation due to their propensity to prevent hooks from being swallowed, instead diverting the hooking location externally or to the mouth (Watson, Epperly and Garrison, et al. 2004, Gilman, Kobayashi, et al. 2007, Sales, et al. 2010). External hooking (e.g., flippers, neck, tail) is considered low risk, as the hook does not affect highly sensitive areas.

There remains some dispute over the advantage of hooking turtles in the mouth rather than the throat, due to the sensitivity of jaw structures and the resilience of the epigastric muscles that line the esophagus of turtles. Once hooked, though, external hooks remain easier to remove, especially if they hook in locations other than the mouth (e.g., flipper). In general, loggerheads are more likely to bite/swallow bait and thus become deep-hooked or hooked in the mouth; leatherbacks are typically externally hooked in a flipper and/or become entangled in leader lines.

When changing hook styles, hook size and turtle size are also factors that warrant consideration. Changing to smaller circle hooks (16/0) from J-hooks is more effective in areas where juvenile turtles are the primary cohort interacting with gear. However, larger turtles are still capable of biting and swallowing the smaller circle hooks, thus making 18/0 and larger circle hooks more effective at preventing turtle bycatch. In most studies, circle hooks displayed >50% reduction of turtle BPUE. Reduction rates were even greater when bait-type was switched from squid to mackerel.

Concerning target catch, circle hooks and mackerel bait receive mixed results. In some fisheries, such as the Western Atlantic NED PLF, target catch was reduced by 86%. In others, the effect on target catch was insignificant with gear changes, with one study actually improving target catch.

Another tool used in PLF predicts hotspots for turtle/fishery interactions by mapping SST gradients. While this tool provides reliable data, high turtle density correlates with high target catch density, thus discouraging fishers from making use of it unless cap limits are being approached. Other methods, such as altering set depth and day versus night setting did affect turtle BPUE. However, both of these aspects are typically set by fishery best practices and thus are not amendable.

In the northwestern Atlantic and a few southwestern Atlantic PLFs, approximately equal numbers of each species are captured as bycatch. Thus, a reduction in bycatch rates by switching to circle hooks with mackerel bait aids both species equally. In the Pacific, where leatherback populations have decreased up to 90%, the majority of turtles caught and internally hooked are loggerheads, and interactions with leatherbacks are exceedingly rare. Thus, while larger numbers of loggerheads are spared by implementing gear changes, the relatively smaller reduction in leatherback bycatch in this region is still significant and vital to the recovery of this population.

The extensive amount of literature that discusses and focuses on longline bycatch is often understood to indicate that PLFs are the dominant fisheries that threaten sea turtle populations. However, the discrepancy in attention is more a factor of data availability. Pelagic longline fisheries receive intense scrutiny due to bycatch characteristics (i.e., sea turtles of higher reproductive value are more heavily impacted), the high-value of the target catch, and the locations of fishing grounds in international waters (Lewison and Crowder 2007). The result of this scrutiny is vast data production when compared to other fisheries. While PLF bycatch is certainly a significant factor, the impacts from other fisheries (i.e., gillnet and trawl fisheries) may be equitable to those of PLFs. Additionally, mortality in these other fisheries is often significantly higher than those reported in longlines. When hooked on shallow lines, turtles can typically still surface for air. In contrast, turtles caught in submerged gillnets and trawl nets are trapped until the gear is recovered, significantly increasing direct mortality once captured.

The global passive set net industry is difficult to regulate and monitor as it largely consists of small-scale and artisanal fisheries. Collectively, their impact can be severe, especially in nearshore operations where seasonally intense fishing pressure along migration corridors and nesting beaches coincides with breeding season. These seasonal migrations can lead to frequent interactions with coastal passive nets (Gass 2006). Impacts from small-scale, nearshore fisheries intensely affect both species during their neritic life stages. For loggerheads, this is the majority of their adult life, but also includes the neritic juvenile stages when they begin recruiting to nearshore environments.

Modifications to leader height in pound net fisheries and overall net height in passive drift gillnet fisheries reduce interactions with loggerheads and leatherbacks, respectively.

In gillnet fisheries, shortening the height of drifting gillnets can reduce the rate of turtle bycatch, though this reduction typically correlates with a reduction of target catch. However, due to the costs associated with net destruction from turtle interactions (especially leatherbacks), the reduction in target catch can be deemed an acceptable cost to some fishers. In pound net fisheries, lowering the leader height is effective at reducing turtle/gear interactions without affecting target catch. Other proposed methods, such as altering marking light color, changing tie-down length, and eliminating buoys showed no significant reduction in turtle bycatch in the reviewed studies.

Trawl fisheries are reported by many authors to have the most deleterious effects on marine turtle populations. To combat the immense number of turtles being taken in trawl fisheries, the NMFS developed TEDs. The creation of the NMFS TED and the enforcement of later editions of the device have drastically reduced these impacts, especially in the U.S. shrimp trawl fisheries. Prior to 2003, TEDs were too small to exclude larger loggerheads and leatherbacks as they were initially designed to exclude only juvenile loggerheads, which were thought to be the primary cohort affected in shrimp trawl fisheries. However, adult loggerheads, as well as sub-adult and adult leatherbacks, also interact with trawl gear. This necessitated the expansion of TED openings, facilitating the exclusion of these larger individuals. Regulations put in place in 2003 increased the minimum height requirements for U.S. TEDs, making them more effective for a broader range of age classes.

In U.S. fisheries, loggerheads are still the dominant species encountered by trawl fisheries, and thus loggerheads constitute the majority of the drastic reduction in trawl bycatch. In other trawl fisheries, such as those throughout the Mediterranean and in the Bay of Bengal, the target catch of larger fish, instead of shrimp, is also often excluded by modern TEDs. Thus, further development of excluder technology is still needed in these regions.

The severe impacts fisheries have on the neritic life stages of loggerheads, along with the sensitivity of population growth rates to survival in these stage, makes them an important target for improved conservation. Large circle hooks with mackerel and TEDs have been shown to aid in reducing bycatch in these age classes. Thus, it is crucial that their implementation spread throughout all fisheries that interact with sea turtles.

Conclusions

Tackling the issue of global marine turtle bycatch is no small feat. It is a multi-faceted issue, affected by variations in seasonal migrations, SST, gear type, target catch characteristics, turtle size, and a host of other factors. Each of these factors requires specialized management techniques that account for fishery-specific characteristics, local resource restrictions, effective enforcement, and the incorporation of stakeholder concerns and proposals. When formulating management strategies, several factors must be considered: local resource potential, feasibility of instituting effective enforcement, severity of threat, potential benefits to species of interest, and effect on target catch, if applicable. Within the two conservation categories discussed here, programs range in scope from local beach monitoring programs to gear regulation in entire fisheries.

This review emphasizes the principle that there is no universal strategy to combat the threats that face each marine turtle species. Nor does there exist a generic fix to apply to threat classes as a whole. Management styles must be tailored to account for all factors that influence marine turtle and human interactions on a local and international scale.

To accomplish this, targeted research is needed that includes input from the people who cause these interactions: fishers, tourism officials, hotel managers, beach maintenance workers, local villagers, and others. The impacts from turtle bycatch are often difficult to conceptualize for individual fishers and smaller fleets. When looking at vessel specific observations, interactions with turtles are rare events, with the majority of sets and trawls interacting with zero turtles. However, due to the sheer size of the global fishing fleet, these few interactions per vessel result in substantial population level impacts. Effectively communicating these concepts to stakeholders through targeted education is a key factor in understanding, and hopefully accepting, the necessity of mandated and suggested conservation strategies.

Further research is needed into improving gear modifications for specific fisheries, including specialized hook shapes and sizes, passive net design, and improved

TEDs. In areas where trawls target larger species, better TEDs need to be designed that effectively exclude turtles while still maintaining target catch quality and quantity.

Population models play a key role in effectively tailoring conservation efforts for local, regional, and global turtle populations. Even with the multitude of management strategies currently enforced, the models presented here still indicate that turtle populations are declining. For successful, robust population recoveries, conservation efforts that affect numerous life-stages, especially both juvenile neritic and adult stages, require expansion. As turtles within these stages interact with longlines, PLF management is further supported as crucial to turtle population recovery. Additionally, research must be continued into understanding our impacts on the different life stages for each population. More accurate survivorship and age class parameters will result in more reliable models to better inform management decisions.

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Appendix I Beach monitoring management strategies.

Species	Region/Beach	Known Increase in Nesting	Threat	Strategy	Summary of Management Outcome	References
Leatherback turle	Mexican Pacific	No	Egg harvest, killing of adult females on the beach and offshore	Nightly monitoring and relocation to hatchery	The project included 4 index and 7 secondary beaches. On index beaches, all clutches were relocated to protected hatcheries. Protection on secondary beaches was inconsistent. Between 1982 and 2004, there has been a massive decline in nesting female numbers despite the release of 270,129 hatchlings during that time.	Martinez et al. 2007
Loggerhead and leatherback	Broward County, Florida	Yes	Beach grooming, erosion	Daily morning surveys and selective nest relocation if erosion or excessive washover is expected	Accounting for cyclical nesting, there has been an overall increasing trend in the total nests laid per year for both loggerhead and leatherback turtles.	Burkholder and Slagle 2015
		Yes	Lighting	Selective caging		

Species	Region/Beach	Known Increase in Nesting	Threat	Strategy	Summary of Management Outcome	References
Loggerhead turtle	Zakynthos Island, Greece	No	Washover, root predation, trampling, tourist/bather disturbance, depredation	Daily monitoring surveys and selective relocation to hatchery if excessive washover was expected; caging of nests in tourist dense areas	Treatments were applied to nests in the 1988-1995 seasons. From 1984 to 2009, all beaches were monitored daily with rare instances of missed surveys. During this 26-year period, there has been no significant increasing or decreasing trend in the annual number of nests. Similarly, neither treatment (relocation or caging) showed a significant increase in hatching success vs. in situ nests.	Kornaraki et al. 2006; Margaritoulis et al. 2011
Leatherback turle	Gandoca Beach, Costa Rica	No	Poaching	Nightly patrols with intense tagging program. Nest treatments were in situ, in situ with tracks camouflaged, relocation of nest to safe zones, relocation of nests to beach hatcheries.	The project lasted from 1990-2004. Prior to the project, poaching claimed almost 100% of eggs laid. During the project, poaching averaged at 15.5% annually. The population remained statistically stable, with a consistent declining trend in nesting numbers since 2000.	Chacon-Chaverri and Eckert 2007

Appendix II Management strategies targeting depredation threats.

Species	Region/Beach	Known Redution in Depredation	Threat	Strategy	Summary of Management Outcome	References
Loggerhead turtle	Florida/Canaveral National Seashore	Yes	Raccoons	Screening of Nests	Prior to implementation of the screening program at Canaveral National Seashore, 90-98% of nests were depredated. Depredation was reduced to 3% with use of screens.	Jordan 1994
Loggerhead turtle	Turkey/Dalyan Beach	Yes	Foxes	Screening of Nests	63% of unprotected nests were depredated vs. 0% of screened nests.	Yerli, et al. 1997
Loggerhead	Zaktynthos Island, Greece	No	Terrestrial Predators	Caging and Relocation to hatchery	There was no correlation between hatching success and either treatment when compared to in situ nests.	Kornaraki, et al. 2006
Loggerhead turtle	Florida/Key Island	Yes	Raccoons	Cages vs Screens	Cages and screens were compared, with partial depredation rates being 0% and 11.4%, and total depredation being 3.6% and 13.6%, respectively.	Addison and Henricy 1994
Loggerhead turtle	Bald Head Island, North Carolina	Yes, with experimental eggs	Foxes	Cages vs Screens	Under normal conditions, depredation was 0% for both treatments and 33% for untreated nests. Under "high motivation" conditions, 0% of caged nests and 25% of screened nests were depredated.	Kurz, et al. (2011)

Species	Region/Beach	Known Redution in Depredation	Threat	Strategy	Summary of Management Outcome	References
Loggerhead turtle	Northern Cyprus, Eastern Mediterranean	Yes, with experimental eggs	Foxes and dogs	Screening of Nests	Only 1 of 36 screened partial predation, 0 were totally predated.	Kinscella, et al. 1997
		No		Chemical deterrents	Dog spray did not show any significant decrease in nest depredation, while mothballs showed moderate deterrance, but were only recommended to be used with screening.	
Loggerhead turtle	Florida/Canaveral National Seashore	Yes	Raccoons	Screening of Nests	2/3 of nests were screened in screening treament areas, resulting in 20-50% reduction in depredation compared to other treatments.	Ratnaswamy, et al. 1997
		No		Predator Removal	50% of raccoon population removed with no reduction in nest depredation.	
		No		Conditioned Taste Aversion	Use of nonlethal estrogen-laced chicken eggs had no significant effect on nest depredation.	
Loggerhead turtle	Florida/ Apalachicola Bay Islands	Yes	Hogs and Raccoons	Predator Control via live traps and spot-and-shoot	During the period between 1990 and 1995, moderate predator control yielded depredation rates of 51% to 100%. With intense predator control in 1996, depredation dropped to 9%.	Bailey, Longiliere and Edmiston 1998

Species	Region/Beach	Known Redution in Depredation	Threat	Strategy	Summary of Management Outcome	References
Loggerhead turtle	Florida/Ten Thousand Islands National Wildlife Refuge	Yes	Raccoons	Live Traps	From 1991-1994, nests experienced 76-100% depredation. After removal of 16 raccoons, nest depredation reduced to 0%.	Garmestrani and Percival 2005
Loggerhead turtle and Leatherback turtle	Florida/Hobe Sound National Wildlife Refuge	Yes	Armadillos and Racccons	Monitoring/ Indexing methodolgy with predator removal	Using blank sand to identify tracks and better target predator removal efforts yielded a reduction in nest depredation from a historical rate of 95% and 42% with an unimproved predator management stratey. down to 28%.	Engeman, et al. 2003
					Further monitoring and indexing method development yielded a decrease in depredation down to 9.4%. The largest impacts were for loggerhead turtles in both studies.	Engeman, et al. 2005
Loggerhead turtle	Florida/Cayo Costa and North Captiva Islands	Yes	Hogs and Raccoons	Monitoring/Indexing methodolgy with predator removal	After one year and two years of management, depredation dropped from 60% and 74% to 0% and 15% for both beaches, respectively.	Engeman, et al. 2010

Appendix III Fisheries management strategies in pelagic longline fisheries.

Species	Fishery	Strategy	Known Reduction in Bycatch	Known Reduction in Target Catch	Summary of Management Outcome	References
Loggerhead Turtle and Leatherback Turtle	Uruguayan pelagic longline fishery	Circle hooks (large-18/0)	No	No	Circle hooks compared vs. American- and Spanish-style longline with J-hooks. Bait-type uncontrolled. The decrease in turtle bycatch for both species was negligible or non-significant. For the majority of target species, however, circle hooks displayed an increase in CPUE.	Domingo, et al. 2012
Loggerhead Turtle and Leatherback Turtle	Brazilian pelagic longline fishery	Circle hooks (large-18/0)	Yes	Varied	Bycatch rates were reduced by 55% and 65% for loggerheads and leatherback, respectively. Catch rates for target species increased for several species, including *Thunnus obesus* and *T. alalunga* , also with several shark species. Capture rates for several species were unaffected, including *T. albacares* and *Coryphanea hippurus* . Only *Xiphias gladius* showed a significant decrease in catch rate, which is a target species.	Sales, et al. 2010
Loggerhead Turtle	Strait of Sicily pelagic longline fishery	Circle hooks (small-16/0)	Yes	No	Circle hooks reduced bycatch rates of immature loggerheads by 70% without a significant effect on target catch (swordfish) CPUE, body size, or weight. All deep-hookings were with J-hooks. The majority of hookings for circle hooks were external.	Piovano, Swimmer and Giacoma 2009

Species	Fishery	Strategy	Known Reduction in Bycatch	Known Reduction in Target Catch	Summary of Management Outcome	References
Loggerhead Turtle	Azores swordfish longline fishery	Circle hooks (small-16/0- and large-18/0)	Yes, but varied	Statistical relevance not given.	Circle hooks resulted in significant reductions in loggerhead CPUE only when compared to Japanese tuna hooks. Hooking location was more heavily impacted by hook style. Rates of deep-hooking of ingested hooks was 60%, 52%, and 13% for J-hooks, Japanese tuna hooks, and all circle hooks, respectively. Within circle hooks, non-offset 16/0 circle hooks had a significantly higher catch rate (especially in the mouth) than 18/0 and offset 16/0 circle hooks.	Bolten and Bjorndal 2005
Loggerhead Turtle and Leatherback Turtle	Portuguese Southern Atlantic swordfish longline fishery	Circle hooks (large-17/0) and mackerel bait	Yes	Yes	The two studies focused compared combinations of J-hooks, a non-offset circle hook and a 10° offset. Circle hooks with mackerel showed a significant decrease in sea turtle by catch compared to J-hook with squid. However, for swordfish, CPUE was significantly higher with conventional J-hook with squid than all other combinations tested.	Santos, et al. 2013, Amorim, et al. 2014

Species	Fishery	Strategy	Known Reduction in Bycatch	Known Reduction in Target Catch	Summary of Management Outcome	References
Loggerhead Turtle and Leatherback Turtle	Western Atlantic Northeast Distant Waters pelagic longline fishery	Circle hooks (large-18/0) and mackerel bait	Yes	Varied	Circle hooks with squid reduced loggerhead and leatherback CPUE by 75% and 74%, respectively. Circle hooks with mackerel reduced turtle CPUE by 91% and 69%, respectively. Circle hooks with squid reduced target catch CPUE of swordfish by 29% and increased that of Bigeye tuna by 23%. Circle hooks with mackerel increased swordfish CPUE by 21% and decreased tuna by 86%.	Watson, et al. 2003, Watson, et al. 2004, Watson et al. 2005
		Day vs. Night	No	No	Variation in CPUE not significant for target or incidental catch.	
		SST Colder vs. Warmer	Yes	No	Dramatic increase in SST of 72° F and 68° F for loggerhead and leatherback, respectively. There was an increase in average catch weight for temperatures below 68° F for swordfish.	

Species	Fishery	Strategy	Known Reduction in Bycatch	Known Reduction in Target Catch	Summary of Management Outcome	References
Loggerhead Turtle and Leatherback Turtle	Hawaiian longline swordfish fishery	Circle hooks, fish bait, seabird bycatch avoidance methods-night-setting and blue-dyed bait	Yes	Varied	Rates for loggerhead and leatherback bycatch were reduced by 90.0% and 82.8%, respectively. Swordfish BPUE increased by 16%. Tuna and combined mahi mahi, opah, and wahoo decreased 50.0% and 34.1%, respectively. Bycatch rates for sharks were also reduced by 36%. Study was not a direct comparison of mitigation methods, but rather a retrospective study that evaluated observed catch rates before and after turtle bycatch regulations were implemented.	Gilman, et al. 2007
Loggerhead Turtle and Leatherback Turtle	Hawaiian pelagic longline Fishery	TurtleWatch	Uncertain	No	Hotspots from this tool seem to have a negligible impact on initial fishing ground choices. However, as interaction limits for the fishery are approached, these maps are more closely followed. Thus, target catch is increased due to preventing the close of the fishery.	Howell, et al. 2008 Howell, et al. 2015

Species	Fishery	Strategy	Known Reduction in Bycatch	Known Reduction in Target Catch	Summary of Management Outcome	References
Loggerhead Turtle and Leatherback Turtle	Western Tropical Pacific longline tuna fishery	Set Depth and Day vs. Night	Yes	N/A	Shallow set hooks (< 100m) during the night had a 0.062 mean CPUE vs. deep sets (150-300m) at night with a 0.012 CPUE for turtles of all species. Variations in setting strategy were based off established methods, thus there were no experimental impacts on target catch.	Secretariat of the Pacific Community 2001
Loggerhead Turtle and Leatherback Turtle	Western Atlantic Northeast Distant Waters pelagic longline fishery	Fleet Communitcation	Yes	No	Bycatch per unit effort was reduced by 50% during the experimental period (2001-2003) as compared to pre-2001 rates.This program occurred in conjunction with the Watson et al. (2004) studies.	Gilman, Dalzell, and Martin 2006
Loggerhead Turtle	Hawaiian pelagic longline fishery	Stealth fishing gear Daytime fishing gear	Uncertain	Yes	There was a 30% reduction in swordfish catch rate for vessel utilizing stealth fishing gear. No turtles were caught with the gear, with predicted turtle catch at 2.7 loggerheads and 0.5 leatherbacks.	Boggs 2003

Species	Fishery	Strategy	Known Reduction in Bycatch	Known Reduction in Target Catch	Summary of Management Outcome	References
Loggerhead Turtle	Western North Pacific shallow-set pelagicl longline fishery	Mackerel vs. squid bait	Yes	N/A	There was a 75% reduction in bycatch rates for loggerheads by switching to mackerel bait vs. squid.	Yokota, Kiyota, and Okamura 2003
		Blue-dyed bait	No	N/A	Bycatch rates were identical for blue-dyed vs. undyed bait for both squid and mackerel.	

Appendix IV Fisheries management strategies in passive net fisheries.

Species	Fishery	Strategy	Known Reduction in Bycatch	Known Reduction in Target Catch	Summary of Management Outcome	References
Leatherback turtle	Trinidad surface drift gillnet fishery	5-15 m mid-water net depth vs. 0-10 m traditional net depth	No	Yes	Lowering the set depth for the nets resulted in 70-75% reduction in target catch with no significant difference is sea turtle bycatch.	Gearhart and Eckert 2007
Leatherback turtle	Trinidad surface drift gillnet fishery	5 m low profile nets vs. 10 m traditional profile nets	Yes	Yes, but acceptable	Leatherback bycatch rates were reduced 11-74%. Target species catch rates were reduced by 35% and 55% for the two target species. The reduction in bycatch was suggested to be offset if longer lengths of net were used. Also, reduction in net costs from fewer turtle interactions made the experimental setup more attractive.	Gearhart, Eckert, and Bergmann 2009
		Trolling vs. traditional gillnets	Yes	Yes, but acceptable	Catch rates were lower for trolling, but CPUE was nearly equivalent. Turtle bycatch was eliminated during the study.	
		Marking lights: red vs. white	No	No	There was no significant difference between using red or white marking lights.	

Species	Fishery	Strategy	Known Reduction in Bycatch	Known Reduction in Target Catch	Summary of Management Outcome	References
Loggerhead turtle	Baja California Sur gillnet fishery	1 m low-profile nets vs. 2 m traditional nets	No	Yes	There was no significant difference in sea turtle bycatch rates. However, there was a significant reduction in target catch with the experimental nets.	Peckham et al. 2009
		0.9 m tie-down length vs. 1.8 m tie-down length	No	No	There was no significant difference in sea turtle bycatch rates. However, there was a significant increase in target catch with the experimental nets.	
		Elimination of buoys on net float line	Yes, but not significant	No	In nets set below 32 m, there was a 47% reduction in sea turtle bycatch, though this was not statistically significant. There was no significant difference in target catch rates. In nets set shallower than 32 m, only one turtle was caught, and thus there was insufficient statistical power. Target catch rates were higher, though insignificantly so.	
Loggerhead and other turtle species	South Japan pound net fishery	Escape hole with self-closing flap	Yes	Yes, but minor	Close to 80% of turtles who migrated into the bag were able to escape through the 40 x 50 cm hole. Retention rate of caught fish was 96%.	Abe and Shiode 2009

Species	Fishery	Strategy	Known Reduction in Bycatch	Known Reduction in Target Catch	Summary of Management Outcome	References
Loggerhead, leatherback, and another turtle species	Chesapeake Bay Virginia pound net fishery	Leader heights reduced to ⅓ vs. traditional leader height	Yes	No	All turtle/gear interactions in the two study years occurred with traditional control leaders with the exception of one leatherback who interacted with the experimental leader. This was the only leatherback encountered in the the study. There was no significant difference in harvest weights between the two leader designs.	Silva, DeAlteris, and Milliken 2011
Loggerhead and other turtle species	North Carolina Pamlico Sound flounder gillnet/pound net fishery	Incidental take permits and partial closure of fishery for gillnets	Yes	Yes	Between 1999 and 2009, the management used evolving standards for incidental take permits and fishery closures. Closures varied from mesh size limits applied against large mesh sizes to spatial closures that applied to all mesh sizes. Loggerhead takes averaged 15.5 throughout the study. Takes were reduced to zero in 2001, 2003, and 2004. Maximum takes were 52 and 65 in 2000 and 2002, respectively. All other years ranged from 4 - 8 takes of loggerheads. Closures negatively affected target catch.	Byrd, Hohn, and Godfrey (2011)