Ecological Risk Assessment of Fisheries on Sea Turtles in the South Western Indian Ocean

Bernice Mellet

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By

Bernice Mellet

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Supervisor: Dr R Nel

Co-Supervisor: Dr L Harris

Declaration by candidate

Name: Bernice Mellet

Student Number: 212464922

Qualification: MSc Zoology

Title of project: Ecological Risk Assessment of Fisheries on Sea Turtles in the South Western Indian Ocean

Declaration:

In accordance with Rule G4.6.3, I hereby declare that the above-mentioned dissertation is my own work and that it has not previously been submitted for assessment to another University or for another qualification.

Signature:

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General Abstract

The SWIO is an area of great biodiversity and included in the diverse species that occupy the region are five species of sea turtles that include green turtles, hawksbills, leatherbacks, loggerheads and olive ridleys. Despite considerable conservation efforts at sea turtle rookeries in the South Western Indian Ocean, only green and loggerhead turtle populations have shown an increase in population size in recent years (<10 years), whereas leatherbacks remained stable and hawksbills and olive ridleys declined. This begs the question if fisheries (or other offshore pressures) are responsible for slowing the recovery of these populations in the region, and if so, which specific fisheries are responsible for this trend? Several offshore (mostly industrial) and coastal (mostly artisanal) fisheries overlap with sea turtle distribution at sea. Industrial fisheries that are globally known to have a demonstrable impact on sea turtle populations are longline and to a lesser extent purse seine fisheries, whilst prawn trawl, gillnet and beach seine fisheries are coastal fisheries with a known negative impact on sea turtle populations. Holistic conservation strategies should be developed that include both land and sea protection for sea turtle species. It is thus necessary to identify and manage offshore threats including fisheries activities, particularly those fisheries that are showing the highest risk to sea turtle populations. This prompted an investigation into the bycatch rates and mortality of all sea turtle species that occur in the SWIO region in several offshore and coastal fisheries including both industrial (longline, purse seine and prawn trawl) and artisanal (including gillnet and beach seine) fisheries. The specific aims were (i) to identify and quantify the interactions (and if possible mortality) of sea turtle species in fisheries and (ii) to identify vulnerable species/populations to fishing operations using a semi-quantitative Ecological Risk Assessment (ERA) in the form of a Productivity-Susceptibility Analysis (PSA). Published information, online databases and technical reports were used as data sources to establish a database containing essential information regarding fishing effort and sea turtle bycatch in the region. The existing information was used to map fisheries extent and effort within the region, and to perform bycatch calculations.

Interactions and mortality rates for sea turtles in five fisheries were quantified using bycatch rates from regional studies. Between 2000 – 2011, Industrial longline and purse seine fisheries captured sea turtles at a rate of 4 388 indiv.y^{-1}, with the mortality rate being 189 indiv.y^{-1}. The bulk of these interactions were in the longline industry that captured 4 129 ± 1 376 indiv.y^{-1}, with a corresponding mortality rate of 167 ± 53 indiv.y^{-1}. The most

commonly caught species (in longlines) were loggerheads and leatherback turtles, but the greatest impact is expected to be on the leatherback population due to the high interaction rate relative to population size. The bycatch (259 ± 34 indiv.y^{-1}) and mortality (20 ± 2 indiv.y^{-1}) rates of sea turtles in the purse seine fishery was considerably lower than the longline fishery. The purse seine fishery thus does not seem to have a significant impact on sea turtle populations in the SWIO. The impact of all forms of fish aggregation devices were excluded from the analysis as the impacts of these are poorly documented.

Coastal prawn trawl, gillnet and beach seine fisheries captured an estimate of between 50 164 - 52 370 indiv.y^{-1} from 2000-2011. The highest bycatch rate was estimated for gillnet fisheries (40 264 indiv.y^{-1}) followed by beach seine fisheries (9 171 indiv.y^{-1}) and prawn trawl fisheries (at 1089 – 2795 indiv.y^{-1}). The gillnet fishery could be responsible for slowing the recovery rate of green turtle and leatherback populations in the SWIO due to the high capture rates in this fishery compared to the population sizes of the species. Beach seine and prawn trawl fisheries are not expected to be hamper the recovery rate of any of the populations in the SWIO due to the low levels of interactions and low mortality rates compared to the population sizes. There are however very few data available regarding the bycatch of sea turtle species within these fisheries, highlighting the need for further research regarding this.

A Productivity-Susceptibility Analysis (PSA) was used to evaluate the relative vulnerability of species to fisheries, and is frequently applied in data poor situations. Limited data on sea turtle life history characteristics and population dynamics of species in the SWIO prompted the use of a PSA to determine the species most vulnerable to fisheries in the region. Results of the PSA indicated that gillnet fisheries poses the largest fishery-related threat to sea turtle populations, specifically the green and leatherback populations. The longline fishery that poses a particular threat to the leatherback population in the SWIO is also a particular concern. A cumulative impact assessment (combining fisheries and other threats) indicated that the SWIO leatherback population is extremely vulnerable to the combination of threats that influence this population in the SWIO.

Even though individual fisheries may pose a small threat, the cumulative impacts of the fisheries can lead to severe impacts on populations such as slowing the recovery rate of populations. There are however significant data gaps that require attention in order to fully

assess the impact of these fisheries on sea turtle populations. Despite the fact that fisheries are not implicated as a mayor reason for the decline in the hawksbill and olive ridley populations in the region, these two species are in decline indicating that there are other factors responsible for the decline not yet identified. It however remains imperative to reduce the mortality from all sources to ensure the continued viability of sea turtle populations in the region.

Keywords: bycatch, sea turtles, Productivity-Susceptibility Analysis, fisheries, mortality, cumulative effects assessment

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Chapter 1
Introduction

Ecosystem-based fisheries management and the application to bycatch species

The world's fisheries contribute greatly to the prosperity and wellbeing of the global community by providing food security, livelihoods to fishers, and income for a substantial portion of the world's population (FAO, 2012). Globally there is a growing demand for animal protein, and fisheries have a greater potential to meet these requirements than terrestrial sources, provided that fisheries follow good governance practices and are managed sustainably (FAO, 2012, Godfray *et al.*, 2010, Pelletier *et al.*, 2011, Pereira *et al.*, 2010). This recognised responsibility has led to a shift from single-species management, that did not take into consideration the importance of maintaining the complexity, structure, and function of ecosystems or the sustainability of fisheries (Garcia *et al.*, 2003b), towards an ecosystems-based approach to fisheries management.

The ecosystems-based approach to fisheries management aims to maintain the integrity of ecosystems to ensure both the sustainability of fisheries and the survival of vulnerable marine life (FAO, 2003, Garcia *et al.*, 2003b). Management organizations thus need to consider the unintended consequences and negative impacts of fisheries operations on ecosystems as part of this approach. These negative impacts include the incidental mortality of non-target species (Lewison *et al.*, 2004a, Livingston *et al.*, 2011), shifts in population dynamics (Hall & Mainprize, 2004), habitat destruction (Thrush *et al.*, 1995) and changes in the structure and function of ecosystems (Livingston *et al.*, 2011, Pikitch *et al.*, 2004). However, there is a limited understanding of the interactions between fisheries and the ecosystems in which these fisheries occur. Consequently, Ecological Risk Assessments (ERAs) were developed to assess the impact of capture fisheries on species, habitats and communities (Gilman *et al.*, 2014a, Gilman *et al.*, 2014b, Hobday *et al.*, 2011, Small *et al.*, 2013).

Ecological Risk Assessments

ERAs are valuable tools that can be used to prioritize issues for fisheries management and research (Fletcher, 2005, Stobutzki *et al.*, 2001), with ERAs evaluating the probability that activities (such as fishing) will cause adverse effects on ecosystem components (Rodier & Norton, 1992). ERAs for fisheries management are based on an exposure-effects approach, because fisheries are deliberate actions, rather than the likelihood-consequence approach that is used for rare or accidental actions such as oil spills (Smith *et al.*, 2007). With the use of ERAs, species for which the risk of negative interaction is the greatest, and areas where these risks are most likely to occur, can be identified (Small *et al.*, 2013). Various types of ERAs exist, including qualitative, semi-quantitative, and quantitative assessments. These assessments are founded on the underlying theoretical basis of a commonly used impact model that is described by the equation:

$$\frac{dB}{dt} = rB\left(1 - \frac{B}{K}\right) - qE \qquad \textbf{Equation 1.1}$$

where B is biomass or population size, r is the intrinsic rate of population increase, K is carrying capacity, q is species catch-ability, and E is fishing effort (Smith *et al.*, 2007).

ERAs can follow a hierarchical approach moving from largely qualitative to quantitative assessments (Fig. 1.1). Qualitative assessments (level 1), based mostly on expert opinion, scan the potential impact that a fishery may have on a host of species (Fletcher, 2005, Hobday *et al.*, 2007). The next level of analysis is semi-quantitative (level 2), and focuses only on species that have medium or high risk. These assessments are more varied, with several different approaches available to assess the impact of an activity on both target and non-target species (Arrizabalaga *et al.*, 2011, Francis, 1992, Lane & Stephenson, 1998, Patrick *et al.*, 2010, Stobutzki *et al.*, 2001), their extinction risk (Cheung *et al.*, 2005, Musick, 1999, Roberts & Hawkins, 1999), and ecosystem viability (Astles *et al.*, 2006, Fletcher, 2005, Jennings *et al.*, 1999). These assessments use data on specific attributes of a species to rank species according to their vulnerability to a specific action or threat (Hobday *et al.*, 2007, Ormseth & Spencer, 2011, Small *et al.*, 2013), usually in the form of a Productivity-Susceptibility Analysis (PSA) to identify the relative risks to species (Arrizabalaga *et al.*, 2011, Milton, 2001, Ormseth & Spencer, 2011, Patrick *et al.*, 2010, Smith *et al.*, 2007). Species identified as high risk, and for which data are available, can be analysed further using quantitative (level 3) model-based analyses (Dulvy *et al.*, 2004, Hobday *et al.*, 2007). After each level of analysis, a risk management response is affected. These may include mandating the use of mitigation measures in fisheries, or applying time-area closures in order to minimize the risks that the fishery poses to a specific entity.

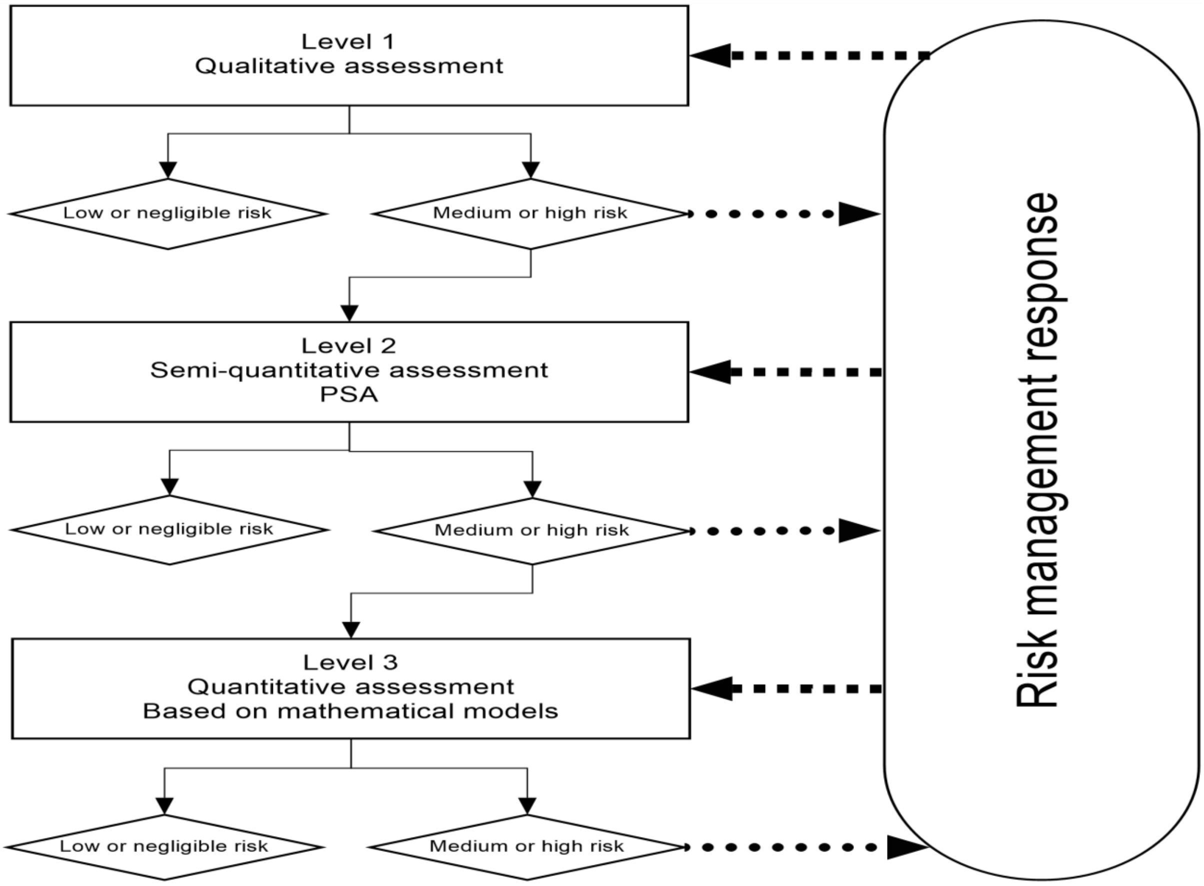


Figure 1.1 Flowchart displaying the hierarchical approach to ecological risk assessments, within the framework of ecosystem-based fisheries management. Square dashed lines indicate reassessment after management measures are implemented, and round dashed lines indicate where management measures are recommended and subsequently implemented (after Hobday *et al*., 2007).

Both qualitative (level 1) and semi-quantitative (level 2) risk assessments use a combination of attributes of both the species or populations and fishery operations to evaluate the potential impacts on target and non-target species (Astles *et al.*, 2006). Both these types of assessments are useful for rapid risk assessment especially where there are data deficiencies in either species or fisheries information (Kirby, 2006). In quantitative assessments (Level 3), complex mathematical models solve the full equation (Eq. 1.1) or equivalents thereof, and these assessments are mostly conducted for the target species of a fishery (Gilman *et al.*, 2014a, Gilman *et al.*, 2014b, Hobday *et al.*, 2011, Small *et al.*, 2013). Quantitative assessments have require large data sets and employ model-based analyses to document absolute population-level effects and to assess the extirpation risk of a population from fishing mortality (Dulvy *et al.*, 2004, Hobday *et al.*, 2011, Kirby, 2006). Furthermore, direct quantitative assessments require high-resolution data on catch rates of species, detailed population demographic information, and knowledge of the spatial-temporal overlap between fishery effort and the distributions of species. These datasets are rarely available for non-target species (Livingston *et al.*, 2011). As an alternative to quantitative assessments, semi-quantitative and qualitative risk assessments are used instead (Pikitch *et al.*, 2004), with semi-quantitative PSAs being the best suited in analyses where data limitations exist.

Productivity-Susceptibility Analysis (PSA)

A PSA expresses the vulnerability of a species or population to a specific fishery, based on the productivity of the species/population in relation to its susceptibility to capture in a fishery (Cortes *et al.*, 2010, Patrick *et al.*, 2010). In these analyses, 'productivity' refers to the intrinsic rate of increase of a species/population, which is determined by factors such as fecundity, natural mortality and age to maturity (Hobday *et al.*, 2011). Productivity thus describes the level of interference a species/population can withstand, as well as the capacity of the species/population to recover from an impact once the impact is alleviated (Hobday *et al.*, 2011). Species with high fecundity and growth rates have a high productivity and can thus sustain high levels of exploitation compared to species with low fecundity and growth rates (Kirby, 2006). 'Susceptibility' describes the potential interaction between a species/population and a specific fishery. The susceptibility is estimated using metrics such as geographic overlap in the distribution of a population and fishing effort, the behaviour of each species (e.g. diving depth vs fishing depth, attraction to fish aggregation devices, etc.), which may affect the likelihood of being captured (Small *et al.*, 2013). In a PSA, species/populations with low susceptibility scores and high productivity scores are ranked as having lower risk (low vulnerability score) and higher capacity to sustain fishing pressures. Conversely, those with low productivity but high susceptibility scores are ranked as vulnerable to fisheries impact, and should be carefully managed (Stobutzki *et al.*, 2001).

Marine vertebrates such as mammals, sea birds, sharks and sea turtles often become entangled or hooked in fishing gear even though they are not the targeted catch (Crowder & Murawski, 1998). The risk assessment of sea birds and sharks have received particular attention in recent years (Arrizabalaga *et al.*, 2011, Cortes *et al.*, 2010, Gallagher *et al.*, 2012, Rowe, 2010, Waugh *et al.*, 2008a, Waugh *et al.*, 2012), however less attention was given to sea turtles and marine mammals (Nel *et al.*, 2013b, Waugh *et al.*, 2008b). Sea turtle species in the SWIO region interact with a variety of fisheries (Bourjea *et al.*, 2008, Brazier *et al.*, 2012, Humber *et al.*, 2011, Kiszka, 2012a, Petersen *et al.*, 2009), however the interactions with these fisheries are not well documented, with sparse information regarding bycatch rates in fisheries. In order to do quantitative assessments of fisheries impacts on sea turtle populations, substantial knowledge is required regarding the population dynamics, catch rate and mortality rates of sea turtles. However, there is a paucity of data on these elements for sea turtle populations, making a quantitative assessment impossible. For the purpose of this dissertation, a PSA will thus be used to perform the ERA.

Sea turtles as focal species

There are seven extant species of sea turtles globally that inhabit nearly all oceans and occupy broad geographical ranges (Wallace *et al.*, 2010a). Five of these species frequent the waters of the Western Indian Ocean. These are green turtles (*Chelonia mydas*), hawksbills (*Eretmochelys imbricata*), leatherbacks (*Dermochelys coriacea*), loggerheads (*Caretta caretta*), and olive ridleys (*Lepidochelys olivacea*). Sea turtles inhabit nearly all oceans and occupy broad geographic ranges (Wallace *et al.*, 2010a). However, their natal philopatry (Bowen *et al.*, 1992) and limited reproductive interactions among regional populations result in limited gene flow within the species, and therefore separate breeding populations/stocks (Bowen *et al.*, 1998, Bowen *et al.*, 1994, Bowen *et al.*, 1992, Dutton *et al.*, 1999, Karl *et al.*, 1992). This has led to the division of global sea turtle populations into regional management units (RMUs) based on their genetic distinctiveness (where available) (Wallace *et al.*, 2010a) or other life-history characteristics (Fig. 1.2). However all species share common characteristics, with some local variation in their life histories (Meylan & Ehrenfeld, 2000, van Buskirk & Crowder, 1994). In order to understand the potential impact of fisheries on these sea turtle populations, it is important to review the life history factors that drive population productivity.

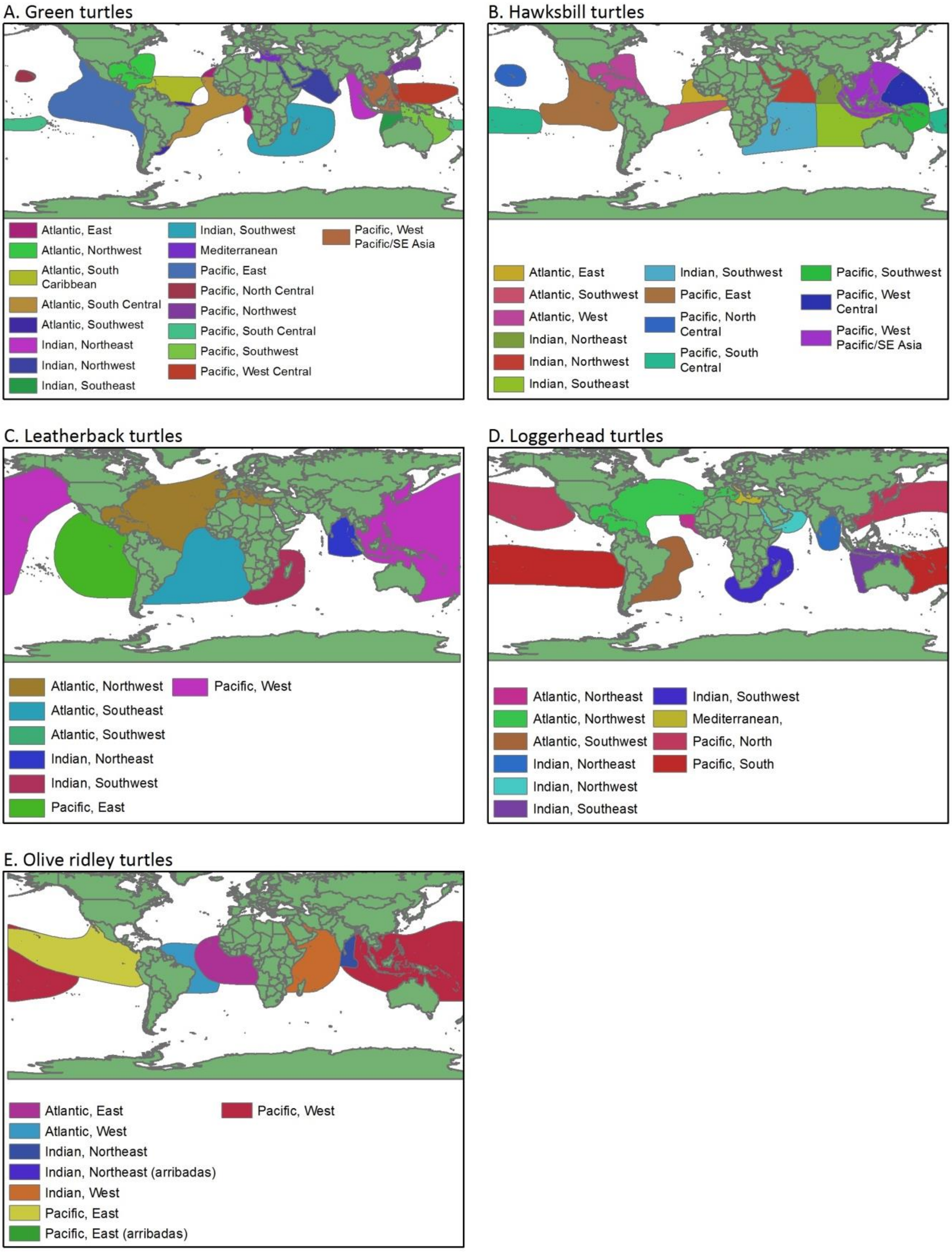


Figure 1.2 Regional management units (Halpin *et al.*, 2009, Wallace *et al.*, 2010a) for green turtles (A), hawksbills (B), leatherbacks (C), loggerheads (D) and olive ridleys (E).

Brief life history of sea turtles

Both male and female sea turtles typically demonstrate natal philopatry to nesting and breeding areas (FitzSimmons *et al.*, 1997a, Limpus *et al.*, 1992); however, males do not restrict their mating efforts to natal breeding grounds. They may copulate with females (sometimes from other regional nesting populations) in coastal feeding habitats or migratory corridors (Bowen *et al.*, 2005, FitzSimmons *et al.*, 1997, Karl *et al.*, 1992). Males and females arrive at the breeding grounds before the nesting season starts. After several incidences of copulation, males depart to foraging areas, whereas females come ashore to nest several times, generally laying between 50 and 130 eggs per nest depending on the individual and species (Bjorndal & Carr, 1989, Broderick *et al.*, 2003, Hirth, 1980, van Buskirk & Crowder, 1994). The number of nests and the interval between successive nesting attempts varies among species (Eckert *et al.*, 2012, Miller, 1997). After nesting, migrating sea turtles often return to the same foraging areas (Broderick *et al.*, 2006, Godley *et al.*, 2002) and once they settle in their foraging area, they move within specific home ranges where they routinely visit specific locations for foraging and resting (Godley *et al.*, 2003, Seminoff & Jones, 2006, Seminoff *et al.*, 2002b, Taquet *et al.*, 2006). Sea turtles initiate post-nesting migrations before hatchlings emerge from the nests, and consequently no parental care is given to offspring.

Nesting may be strictly seasonal or occur throughout the year, especially at nesting sites closer to the equator, with marked peaks in nesting activity. Nesting that takes place at more temperate sites occurs during summer (Dodd, 1988, Eckert *et al.*, 2012, Hirth, 1980, Hirth, 1997). Eggs incubate for approximately two months, during which several environmental factors influence the hatching and emergence success of nests, including gas exchange, moisture and temperature (Maloney *et al.*, 1990, Mortimer, 1990). Clutches generally have high hatching success unless external factors such as predation and environmental changes (i.e. rise in nest temperature or inundation of nests) interfere (Magnuson *et al.*, 1990). The sex of hatchlings in the nests is determined by the temperatures experienced during incubation. Females are produced at higher temperatures and males at cooler temperatures (Ackerman *et al.*, 1997).

After hatching, hatchlings rely on their kin using socially facilitated digging to reach the sand surface (Carr & Hirth, 1961). Hatchlings crawl to the ocean where they will swim for several days before reaching currents that will move them away from the coast (Hughes, 1974a, Scott *et al.*, 2011). For approximately ten years, these post-hatchlings will drift in pelagic waters feeding on planktonic prey (Hughes, 1974c, Luschi *et al.*, 2003a). Thereafter they recruit to neritic feeding habitats as developed juveniles where they will remain as sub-adults and adults (Musick & Limpus, 1997) except for leatherbacks that are considered to remain pelagic throughout their lives (Eckert *et al.*, 1989a).

Sea turtles are late-maturing animals and the age to sexual maturity differs among species, populations and individuals (Chaloupka *et al.*, 2004, Tucek *et al.*, 2014, Zug & Parham, 1996, Zug *et al.*, 2002). Once sea turtles reach sexual maturity they will periodically migrate to their natal beaches to breed (Limpus *et al.*, 1992, Musick & Limpus, 1997). Females do not breed annually (Dodd, 1988, Eckert *et al.*, 2012, Hirth, 1980, Richardson *et al.*, 1999), whereas males have shorter remigration intervals, often breeding every year (James *et al.*, 2005a, Limpus, 1993, Wibbels *et al.*, 1990). The remigration interval is this period between successive nesting seasons. Mean remigration intervals vary among species and range from 1 - 9 years (Limpus *et al.*, 1992, van Buskirk & Crowder, 1994).

Sea turtles consequently have a complex life-history strategy. They exhibit several characteristics of K-strategists, such as large body size and high survival of sub-adult and adults compared to hatchlings and juveniles (Heppell *et al.*, 2003b). However, they also produce many small offspring that is characteristic of r-strategists. Given that they display elements of both r- and K-strategies, and particularly the following four characteristics: late maturation; high offspring production; no parental care; and high juvenile mortality, their life-history strategy is thus considered “periodic” (Winemiller & Rose, 1992).

Land-based life-history stages of sea turtles are much more accessible to researchers than those that occur exclusively in the marine environment. Because of this, the stages occurring in the terrestrial environment are most well studied. Monitoring the behaviour and ecology of the species in the marine environment is more complicated. However, understanding their behaviour, ecology and threats that these species face in both their land based and marine environments are essential in order to conserve these species effectively.

Threats to sea turtles

Sea turtles were historically very abundant, however in recent times their numbers have decreased substantially due to human interference and direct exploitation (Mortimer, 2007). All species are now listed as threatened or endangered (IUCN, 2014). Human activities are directly influencing all life-history stages as well as indirectly by impacting upon the associated habitats that each of the life-history stages occupy. Pressures range from land based threats such as egg harvesting, direct take of females and loss of nesting beaches from inappropriate coastal development and sea-level rise (Fish *et al.*, 2008) to incidental capture in small-scale or artisanal coastal fishing operations that are frequently unselective fisheries, and large-scale commercial fishing operations operating mainly on the high seas (Dutton & Squires, 2008).

Land-based threats

Land-based threats to sea turtles come from a variety of natural and non-natural sources. These include egg poaching, harvesting of females, predation, erosion, artificial lighting, beach nourishment, increasing foot and vehicle traffic, beach armouring, sand mining, and global climate change (Lutcavage *et al.*, 1997). Harvesting adult female sea turtles and eggs at nesting beaches for commercial and subsistence purposes occurs worldwide, and is the primary cause of population declines (Chaloupka, 2003). The predation of eggs hatchlings and nesting females by feral pigs, dogs and other species also remains a significant cause of declines in populations in certain areas (Hitipeuw *et al.*, 2007, Kinch *et al.*, 2006, Wurlianty & Hitipeuw, 2006). The encroachment of humans on coastal habitats further contributes to the declines in population abundance by degrading nesting beaches by activities such as sand mining, dredging, beach armouring, beach erosion, pollution, and excessive use of artificial light (Lutcavage *et al.*, 1997). Off-road vehicles on beaches can crush eggs and hatchlings and tyre ruts can become traps resulting in exhaustion of hatchling or increased predation (Hosier *et al.*, 1981, Lamont *et al.*, 2002). Beach erosion from relative sea-level rise is an increasing threat to incubating sea turtle nests (Dutton *et al.*, 2005, Hitipeuw *et al.*, 2007), especially on beaches constrained by seawalls and development (Fish *et al.*, 2008) and beaches with low elevations (Fish *et al.*, 2005). Increasing sand temperatures at some nesting beaches due to global climate change and alterations to beach vegetation may be exceeding the thermal tolerance of embryos resulting in embryo mortality (Fuentes *et al.*, 2011, Hawkes *et al.*, 2007, Hays *et al.*, 2003, Hays *et al.*, 2010, Pike, 2013).

Threats at sea

Threats to sea turtles in their marine environment can indirectly affect sea turtle populations and include habitat destruction or pollution, whilst those that can directly affect them include factors such as incidental bycatch in fisheries and targeted catches. The anthropogenic impacts at sea include boat strikes, dredge and fill activities, underwater explosions, marine debris, plastic pollution (pesticides, oil, and other chemical toxins released from pipelines or accidental spills from ships) and fisheries by-catch, targeted catches of sea turtles and, entanglement in debris (Lutcavage *et al.*, 1997). Vessel traffic resulting in boat strikes is an important cause of sea turtle mortality in some areas, although some of the strikes may occur post-mortem (Magnuson *et al.*, 1990). Dredge activities such as filling for land reclamation may result in sea turtle entanglement in dredging gear (Dickerson *et al.*, 2004, Slay & Richardson, 1988), whilst oil exploration activities and military manoeuvres that cause underwater explosions may lead to death of sea turtles (Magnuson *et al.*, 1990). Pollution, in both solid and chemical form, is another key threat to sea turtles in the ocean. For example, marine debris, including discarded and lost fishing gear and plastics, cause injury and mortality of sea turtles when they ingest this debris or become entangled in it (Carr, 1987a, Schuyler *et al.*, 2014a, Schuyler *et al.*, 2014b). Furthermore, contaminants such as phthalates from plastics, that have toxicological effects and effects on reproduction, have been found in leatherback egg yolks (Cerón *et al.*, 2000, Guerranti *et al.*, 2014). Similarly, recent studies have shown that persistent organic pollutant such as polychlorinated biphenyls (PCBs) and pesticides such as DDT may impact sea turtle populations by modulating immune responses (Camacho *et al.*, 2014, Keller *et al.*, 2006, Lazar *et al.*, 2011, Swarthout *et al.*, 2010).

A variety of fisheries have been implicated in the bycatch of sea turtles including trawls (Henwood & Stuntz, 1987, Lutcavage & Lutz, 1991), purse seines, dredges, pots and traps (Chuenpagdee *et al.*, 2003), longlines (Alfaro Shigueto *et al.*, 2007, Lewison & Crowder, 2007), and gillnets (Casale, 2008, Wallace *et al.*, 2013b). Mortalities associated with coastal fisheries have received much less attention than those in industrial offshore fisheries, however in some cases these inshore fisheries have been shown to have a much greater impact (Alfaro Shigueto *et al.*, 2007, Humber *et al.*, 2011). Targeted catches of sea turtles in mostly coastal fisheries in the SWIO continue to occur regardless of the fact that these are mostly illegal throughout the region (Bourjea *et al.*, 2008).

Incidental captures of sea turtles have expanded in recent years as there has been a substantial expansion of fishing activities in the second half of the twentieth century (FAO, 2005, Lewison *et al.*, 2004a). This is of concern because high by-catch rates can have significant impacts on sea turtle populations (Finkbeiner *et al.*, 2011, Lewison *et al.*, 2004a, Soykan *et al.*, 2008), and has even been implicated as a key factor pushing some populations close to extinction (Spotila *et al.*, 2000). However, the relative catch and extent of impact to sea turtle populations are not equal per unit effort amongst fisheries that incidentally capture sea turtles (Wallace *et al.*, 2013b). The extent of impact depends on the rate of mortality after accidental capture, the life stage of the sea turtles removed by capture, or the size/status of the affected sea turtle population. A number of sea turtle populations are declining worldwide due to the number of threats, making the protection of these species and fisheries mitigation a very high priority.

Conservation Status and Efforts

Due to the significant threats that sea turtles face, the IUCN list all seven species as threatened (IUCN, 2014). The current IUCN Red List of Threatened Species framework does not assess the variation in the status and trend of individual populations of wide ranging species (Godfrey & Godley, 2008, Seminoff & Shanker, 2008). It is essential to assess the threat status of individual populations in order to build towards a sound conservation strategy for each of these subpopulations (Wallace *et al.*, 2010a). All sea turtle species occurring in the South Western Indian Ocean are classified as Endangered, Critically Endangered or Vulnerable by the IUCN Red List of Endangered Species (Table 1.1, IUCN, 2014). Leatherbacks are listed globally as Vulnerable however regionally this species is listed as Critically Endangered (Wallace *et al.*, 2013a).

Table 1.1 Red list status of sea turtle species occurring in the South Western Indian Ocean (IUCN, 2014, Wallace *et al.*, 2013a).

Species	**Red List Status**
Green turtles (*Chelonia mydas*)	Endangered
Hawksbills (*Eretmochelys imbricata*)	Critically endangered
Loggerheads (*Caretta caretta*)	Endangered
Leatherbacks (*Dermochelys coriacea*)	Critically endangered
Olive ridleys (*Lepidochelys olivacea*)	Vulnerable

Even though sea turtles are at risk both on land and in their marine environment, the terrestrial stages in their lifecycle offer unique opportunities for conservation. This is because conservation efforts at nesting beaches offer protection to all land-based life-history stages of sea turtles (eggs, hatchlings, and nesting females) and to the critical habitat supporting breeding (e.g., controlling abundance of non-natural predators, mitigating the effects of artificial lighting on the beach, and restricting inappropriate coastal development). These conservation efforts have been successful in the recovery of sea turtle populations around the world, such as those nesting at Hawaii (Balazs & Chaloupka, 2004), Florida (Stewart *et al.*, 2011), Mexico (Garcia *et al.*, 2003a), Caribbean (Dutton *et al.*, 2005), French Guiana (Fossette *et al.*, 2008b), South Africa (Nel *et al.*, 2013a), Europa and Grande Glorieuse (Lauret-Stepler *et al.*, 2007), and Aldabra (Mortimer, 1985) and Cousin Island (Seychelles, Wood, 1986). However, despite the conservation efforts at nesting beaches some populations are still declining (Lewison & Crowder, 2007). This trend can be attributed to the continued decrease in the number of reproductively mature individuals (Crouse *et al.*, 1987, Frazer, 1992, Heppell *et al.*, 1996b) due to impacts that occur offshore (Finkbeiner *et al.*, 2011, Lewison & Crowder, 2007, Nel *et al.*, 2013a, Spotila *et al.*, 2000).

Even though it is more difficult to conserve sea turtles in the marine environment, successful mitigation measures against fisheries impacts are possible. Mitigation measures at sea include technological changes to fishing gears and time/area closures of fishing grounds. These measures are specifically aimed at reducing sea turtle bycatch and the resultant mortality (Dryden *et al.*, 2008). Technological changes include the use of Turtle Excluder Devices (TEDs) on trawling gear and the use of circle hooks in the longline fishery (Epperly, 2003). Fishery closures of areas during peak times of sea turtle abundance ensure that fisheries operate in areas where there is a lower probability of interaction with sea turtles. These fisheries closures do not cover the entire range of a species (Dryden *et al.*, 2008), but fulfil an important role at smaller scales to protect foraging and inter-nesting habitats (Dobbs *et al.*, 2007, Dryden *et al.*, 2008). The mitigation measures are often however limited to specific geographical regions and implementation of regulations with respect to technological mitigation measures are often lacking. In order to assess the need for these in the South Western Indian Ocean (SWIO) it is necessary to assess the possible impact that fisheries may have on sea turtle population in the region as well as identifying areas where the impact may be the highest.

Aims and Objectives

Even though conservation efforts at nesting beaches are shown to be successful in the recovery of some populations of sea turtles, there are still cases in which populations are not recovering at satisfactory rates (Nel *et al.*, 2013a) and population growth rates for most of the recovering populations remain low (Lauret-Stepler *et al.*, 2007). It is thus necessary to focus our attention on the anthropogenic impacts in the marine environment, such as fisheries by-catch. This will make it possible to determine the fisheries that have the greatest impact on sea turtle populations and to identify possible mitigation measures that can increase the sustainability of these fisheries within an ecosystems approach to fisheries management. In the SWIO, there is significant overlap between the distribution of several sea turtle populations and a variety of fisheries that have the potential to affect sea turtle populations by increasing non-natural mortality rates. The overlap of multiple species with a variety of gear types in the SWIO offers an excellent opportunity to compare relative impacts to species among fisheries. The aim of this dissertation is thus to compare the risk of sea turtles interacting with different fisheries in the SWIO region. In order to do this comparison, the dissertation is divided into a series of discrete chapters according to the research objectives to determine specific fisheries impacts as follows:

- Describe the study area including the distribution and productivity criteria of sea turtle populations that nest within the SWIO region (Chapter 2)
- Quantify how many sea turtles, per species, are captured as by-catch in the large-scale longline and purse-seine fishing operations in the SWIO (Chapter 3)
- Quantify how many sea turtles, per species, are captured as bycatch in coastal prawn/shrimp trawling, gillnetting and beach-seine fishing operations in the SWIO (Chapter 4)
- Assess the vulnerability of sea turtles in the SWIO to these fisheries by:
  - Developing a PSA methodology to assess the vulnerability of sea turtle populations to fisheries (Chapter 5)
  - Assessing the relative productivity of sea turtle population in the SWIO (Chapter 5)
  - Assessing the relative susceptibility of each sea turtle population to capture in large scale (industrial purse seine and longline) fisheries and small scale (prawn/shrimp trawl, gillnet and beach seine) fisheries (Chapter 5)

    - Assessing the vulnerability of sea turtle populations to both large scale fisheries and small scale fisheries in the SWIO (Chapter 5)
    - Comparing the relative threats posed by the various fisheries to determine the fisheries with the potential to cause the highest impact on sea turtle populations (Chapter 5)
- Conclude and summarise the research (Chapter 6)

Chapter 2

The South Western Indian Ocean: its sea turtles and fisheries

Study Area: The South Western Indian Ocean Region

The South Western Indian Ocean (SWIO) is an area of great biodiversity with many different habitat types, dynamic currents, and monsoons. The habitat types occurring in the region include coral reefs, mangrove forests, sea grass meadows, estuaries and wetlands, and offshore banks and seamounts. It is delineated in this dissertation (Fig. 2.1) as the region between the Horn of Africa, Somalia (10 ºN) in the north to approximately as far south as the Prince Edward Islands (46 ºS). The western boundary extends along the coastline of the African continent to Cape Agulhas, and the eastern boundary is partially at 65 ºE and 80 ºE (Fig. 2.1). The landmasses of the SWIO comprises six islands and archipelagos including the island nations Comoros, Madagascar, French Scattered Islands (FSI), Mauritius, Reunion, Seychelles and the British Indian Ocean Territory (or the Chagos Archipelago). Five continental countries are included; these are Kenya, Mozambique, Somalia, South Africa, and Tanzania along the east coast of Africa (Fig. 2.1). There are a few smaller islets (e.g. Bassas de India) and atolls scattered throughout the SWIO that adds to the complexity of the sea floor and resultant coastal habitats.

There are two Large Marine Ecosystems (LMEs) in the region: the Agulhas Current LME towards the south, and the Somali Coastal Current LME to the north (Fig. 2.1), with the Mascarene Plateau occurring to the east of these two LMEs. The Agulhas Current LME, a moderately productive ecosystem (Heileman *et al.*, 2009) extends from 10 ºS to 37 ºS; it covers a large area of southern African waters off the coasts of Mozambique and South Africa and encompasses several islands. The currents and eddies in the LME are influenced by the topography of the seafloor. The Somali Current LME extends from 10 ºS to 11 ºN (Alexander, 1998) and is a moderately productive ecosystem (Heileman & Scott, 2009), with primary productivity being driven by persistent, monsoon-driven upwelling. The migration of the intertropical convergence zone drives the currents in the Somali Current LME. The two main seasons are the northeast (NE) monsoon and the southwest (SW) monsoon, with productivity being the highest during the SW monsoon.

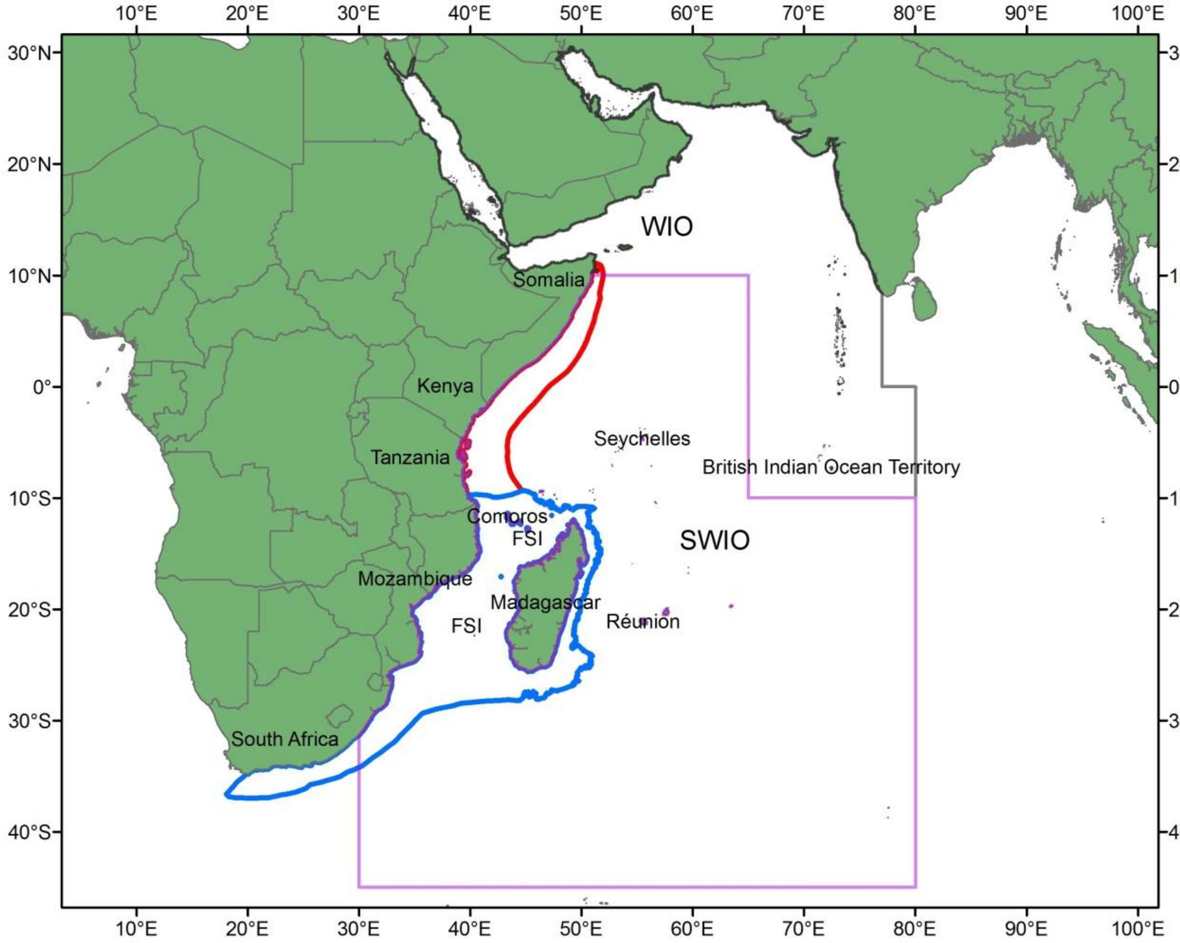


Figure 2.1 Map of the study area showing the two Large Marine Ecosystems (Somali Current LME in red and the Agulhas Current LME in blue) and the boundary of the South Western Indian Ocean (SWIO, in purple) nested within the Western Indian Ocean (WIO, in black).

Several currents in the region determine the fine-scale oceanography, the most prominent of which are the Agulhas Current (AC), East Africa Coastal Current (EACC), East Madagascar Current (EMC), Somali Current (SC), and the South Equatorial Current (SEC; Fig. 2.2). The EMC is responsible for upwelling events (Quartly *et al.*, 2005) and has been observed to retroflect (Quartly & Srokosz, 2004). Anticyclonic eddies form in the north of the Mozambican channel and move southward where they merge with cyclonic and anticyclonic contra-rotating eddy pairs or dipole eddies that originate to the south of Madagascar. The eddies move southwest to feed into the AC (Ridderinkhof & De Ruijter, 2003, Schouten *et al.*, 2003, Tew-Kai & Marsac, 2009).

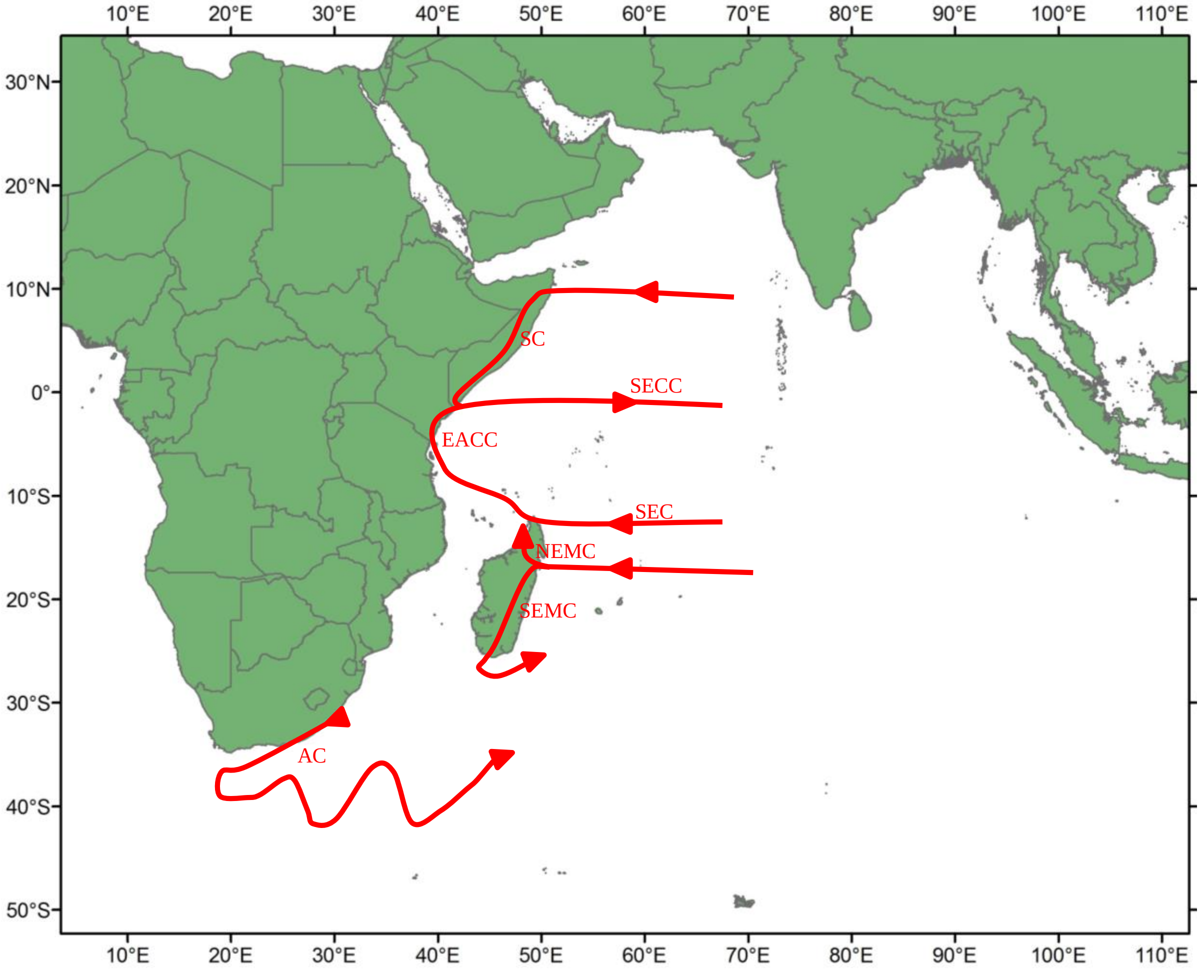


Figure 2.2 Schematic representation of the near-surface currents in the SWIO region during the Northeast monsoon, showing the South Equatorial current (SEC), South Equatorial Counter Current (SECC), Northeast Madagascar Current (NEMC), Southeast Madagascar Current (SEMC), East African Coastal Current (EACC), Somali Current (SC), Agulhas Current (AC). Following ASCLME/SWIOFP (2012).

Source water of the warm AC derives from the East Madagascar Current and eddies from the Mozambique Channel (De Ruijter *et al.*, 2005, Heileman *et al.*, 2009, Siedler *et al.*, 2009). The current originates between 25 ºS and 30 ºS and flows southward following the shelf edge along the East coast of South Africa, at 36 ºS it moves further offshore and follows the contours of the Agulhas Banks (Bryden *et al.*, 2005). At Approximately 40 ºS, south of the Cape of Good Hope, the AC retroflects and majority of the current volume travels eastward to form the Agulhas Return Current that flows along the subtropical convergence (De Ruijter *et al.*, 2005, Heileman *et al.*, 2009, Lutjeharms, 2007).

During the NE monsoon, the SC flows in a southwesterly direction meeting the northward flowing East African Coastal Current (EACC) (Fig 2.2). It flows eastwards at 0-2 ^{o}N into the south equatorial counter current (SECC). When the season changes to the SW monsoon (April to June), the EACC strengthens and the SC changes direction to flow towards the northeast, driven by the prevailing winds. By August the EACC and the SC form a continuous north-eastward flowing current to the East Arabian Current (Heileman & Scott, 2009). The circulation in the Mozambican channel is also affected by the monsoon seasons, with the southwest monsoon associated with strong winds and high volume transport through the Mozambique Channel, while the NE monsoon is associated with low winds and very low levels of transport through the channel (Biastoch & Krauss, 1999). NE monsoon winds have also been correlated with upwelling off the coast of Mozambique (Malauene *et al.*, 2014). During the SE monsoon, wind induced coastal upwelling brings cold, nutrient rich water to the surface layer creating favorable conditions for the fisheries (Belkin & Cornillon, 2007, Heileman & Scott, 2009).

Sea Turtles of the SWIO

There are five species of sea turtles that occur within the SWIO region, green turtles (*Chelonia mydas*), hawksbill turtles (*Eretmochelys imbricata*), leatherback turtles (*Dermochelys coriacea*), loggerhead turtles (*Caretta caretta*), and olive ridley turtles *(Lepidochelys olivacea*). All five species are listed as threatened by the IUCN: leatherbacks and olive ridleys are vulnerable; green turtles and loggerheads are endangered; and hawksbills are critically endangered (IUCN, 2014), regionally however leatherback turtles in the SWIO are critically endangered (Wallace *et al.*, 2013a). Several characteristics influence the demographics and consequently the productivity of sea turtles. These characteristics include population size, genetic distinctiveness of populations, fecundity, and natural mortality. The susceptibility of sea turtles to fishing operations are influenced by several factors, such as the overlap between sea turtles and fisheries (and thus distribution of sea turtles influences their susceptibility), the behaviour of the species (e.g., feeding and diving behaviour), and the probability of capture (the rate of bycatch). The productivity characteristics are described in detail for each species in the following sections highlighting rates that are specific to populations in the SWIO. Throughout the dissertation, Regional Management Units (RMUs) as per Wallace *et al.* (2010a) (See Chapter 1 for details) will be treated as the unit for all analyses.

Green turtles (*Chelonia mydas*)

Green turtles are the most abundant species in the region, estimated at >10 000 females nesting annually (Appendix A). Nesting sites for green turtles are primarily located on isolated islands including Comoros, Europa, Glorieuses, Mayotte, Seychelles, and Tromelin, and along the continental beaches of Kenya, Madagascar, Mozambique, and Tanzania (Fig. 2.3). There has been a decrease in the long-term trend of annual nesting numbers at Europa (Le Gall *et al.*, 1986, Lebeau *et al.*, 1979, MTSG, 2004, Rene & Roos, 1996) and Tromelin (MTSG, 2004, Rene & Roos, 1996), however there has been a recent increase (over the past 10 years) at Europa (Lauret-Stepler *et al.*, 2007), Glorieuses (Lauret-Stepler *et al.*, 2007), Mayotte (Bourjea *et al.*, 2007a), Seychelles (Aldabra) (Mortimer *et al.*, 2011b) and Comoros (Moheli) (Bourjea, 2012). Three genetic stocks have been identified for green turtles in the SWIO (Bourjea *et al.*, 2007b), however all nesting aggregations within the region will still be treated as one management unit due to the proximity of the rookeries and because these stocks overlap substantially on the feeding grounds (Bourjea *et al.*, 2013), thus facing similar threats throughout their distributions. Satellite tracking of predominantly post-nesting female green turtles (Fig. 2.4, Appendix B) has shown that females nesting along the east African seaboard confine their migrations to the continental coast; this is in contrast to populations nesting on islands. These island nesters migrate towards the east African coast and Madagascar, to use shallow waters and sea grass beds as feeding grounds (Bourjea *et al.*, 2013).

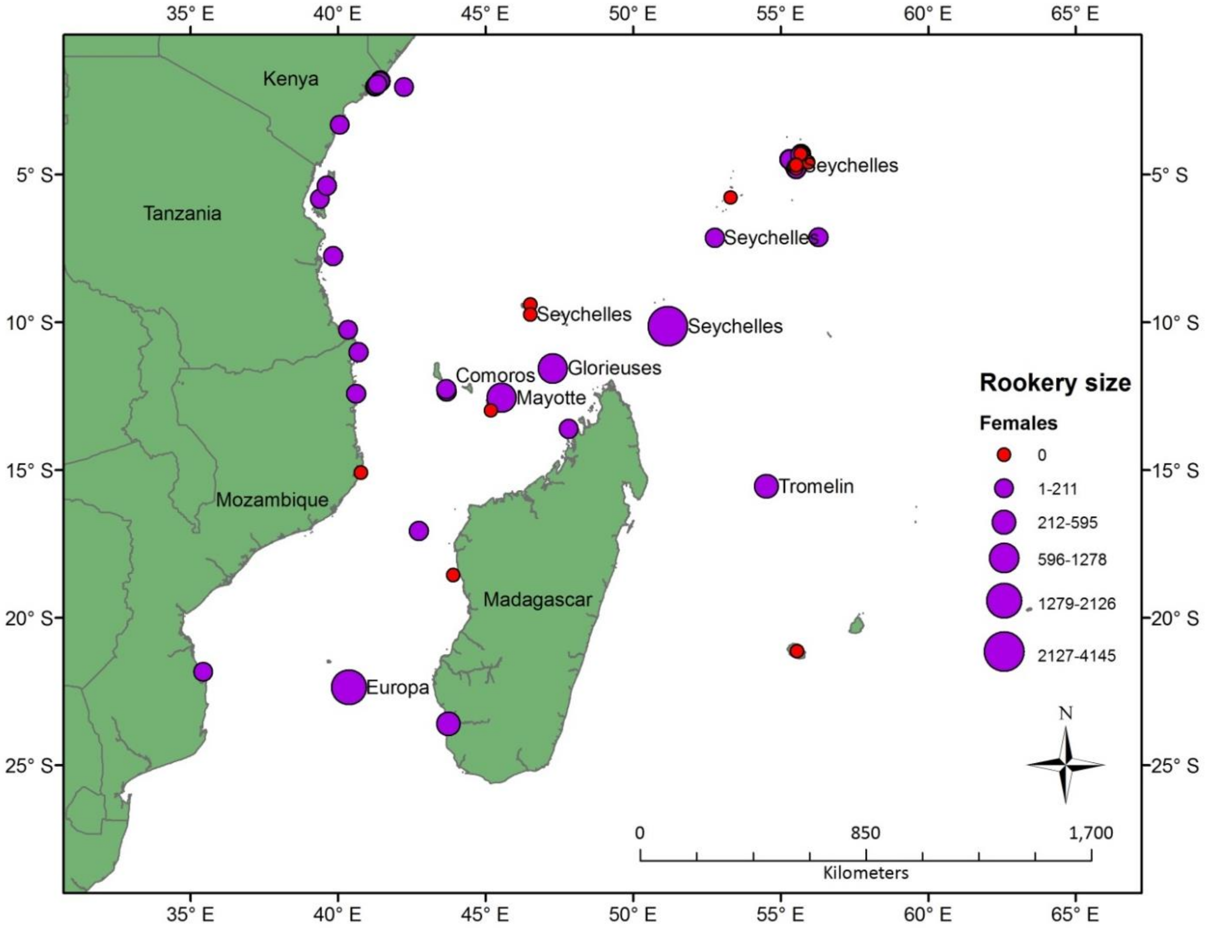


Figure 2.3 Distribution of green turtle nesting sites (rookeries) in the SWIO as per the OBIS/SEAMAP online database (Kot *et al.*, 2013) supplemented with additional published literature on nesting areas (Appendix A). Rookery sizes are reported as number of nesting females per annum. Rookeries with 'zero' females (shown in red) are rookeries where nesting has been confirmed but the rookery sizes are currently unquantified.

It takes between 20 – 40 years for green turtles to reach sexual maturity (Chaloupka *et al.*, 2004, Limpus & Chaloupka, 1997, Seminoff *et al.*, 2002c, Zug *et al.*, 2002, Zug & Glor, 1998), however no studies have been conducted on their age to maturity in the study region. This research gap notwithstanding, age to maturity for SWIO green turtles is estimated at 33.3 years (MTSG, 2004), The reproductive longevity of these sea turtles varies from 17 – 23 years (Chaloupka *et al.*, 2004, Fitzsimmons *et al.*, 1995, Hirth, 1997, Vera, 2007). From these estimates, the maximum reproductive age for green turtles is suggested to be from 50.3 – 56.3 years, and generation length is estimated at 41.8 – 44.8 years (where generation length is the age at half of the maximum reproductive lifespan). Once green turtles reach sexual maturity, they migrate between nesting and feeding grounds every 2.6 ± 1.1 (SD) – 3 ± 1 (SD) years (Anastácio *et al.*, 2014, Bourjea *et al.*, 2007a), laying 112.01 ± 23.22 (SD) – 152 (range 115-197) eggs per clutch (Anastácio *et al.*, 2014, Garnier *et al.*, 2012, Hughes, 1974c, Innocenzi *et al.*, West *et al.*, 2013), with females laying between 2.5 ± 1.2 (SD) – 3.5 clutches per season (Anastácio *et al.*, 2014, Bourjea *et al.*, 2007a, Garnier *et al.*, 2012, Le Gall, 1988, West *et al.*, 2013). In the SWIO, hatching success ranged between 71 – 87 ± 19 (SD) % (Anastácio *et al.*, 2014, Garnier *et al.*, 2012, Innocenzi *et al.*, West *et al.*, 2013), with emergence success ranging between 64.8 ± 29 (SD) – 84.5 ± 20.4 (SD) % (Anastácio *et al.*, 2014, Garnier *et al.*, 2012, Innocenzi *et al.*).

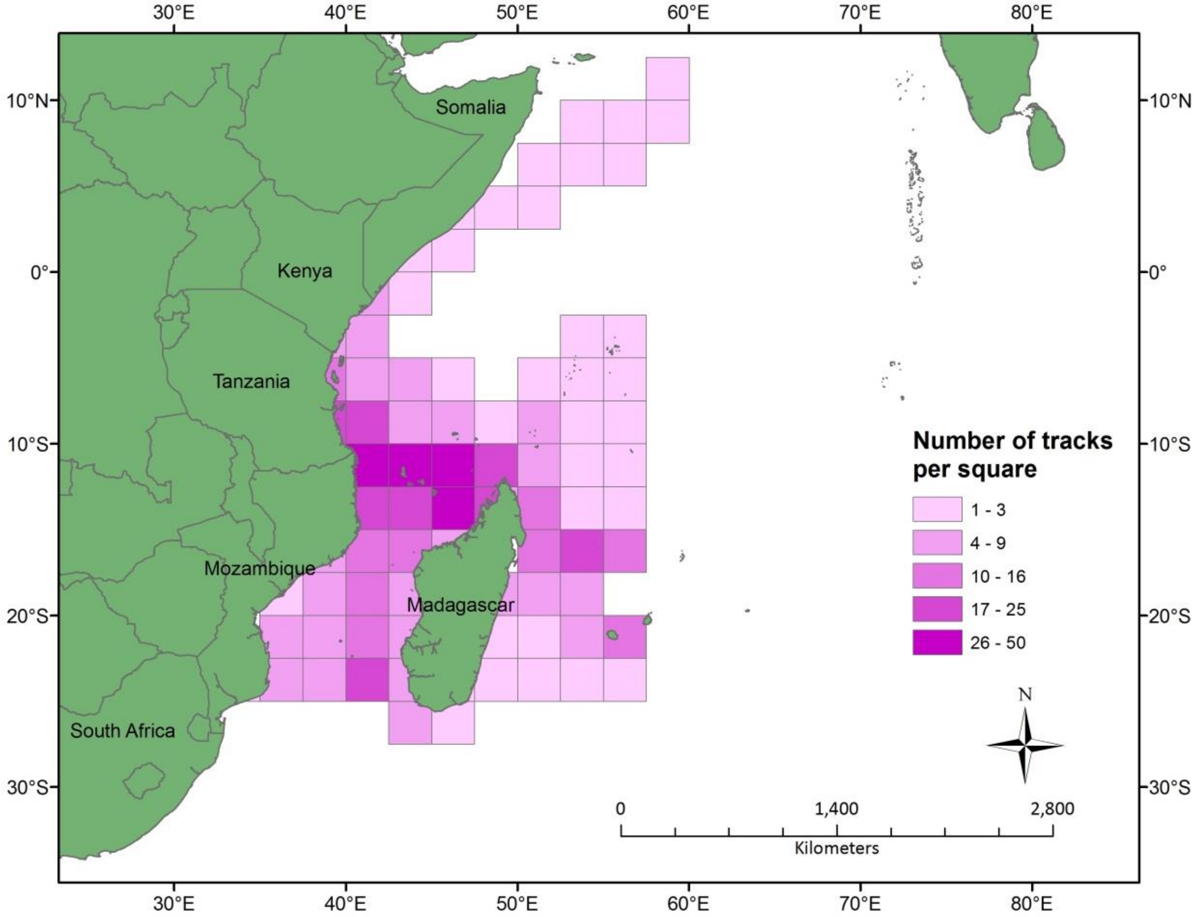


Figure 2.4 Approximate distribution of (mostly) post-nesting green turtles in the SWIO generated from published and unpublished satellite tracking studies (Appendix B). Densities are shown as number of tracks per 2.5° x 2.5° square.

Young green turtles occupy pelagic habitats. They are often associated with sargassum rafts (Carr, 1987b) but have an omnivorous diet (Bjorndal, 1985, Boyle & Limpus, 2008, Salmon *et al.*, 2004). When these juveniles leave the pelagic habitat and move to benthic feeding areas, their diet shifts towards a herbivorous diet that they will follow for the rest of their lives (Carrion-Cortez *et al.*, 2010, Hatase *et al.*, 2006, Heithaus *et al.*, 2002, Mortimer, 1981, Seminoff *et al.*, 2002a). It appears that this is not an entirely strict feeding strategy, because adults of some populations of green turtles have been shown to forage in oceanic waters (Parker *et al.*, 2011).

Diving depths attained and habitats occupied by green turtles vary amongst age classes (Table 2.1; see references therein). The mean diving depth of post hatchlings is 2.5 m, with a maximum diving depth of 9.3 m in their oceanic developmental habitat. For juveniles in the neritic developmental habitat, mean diving depth varies from 2.1 – 21.7 m, with a maximum of 37.1 m. However, juvenile green turtles have been recorded to dive up to 91 m in the oceanic developmental habitat. Diving depths continue to deepen into the adult phase; in the neritic and oceanic inter-nesting habitats, dives range 2.8 - 20.5 m, but can reach up to 65.2 m. The deepest dives are attained during migrations, with the average diving depth ranging 28.7 – 38.8 m, but can be more than four times deeper at 138 m.

Table 2.1 Mean and maximum dive depths (in meters) of green turtles across various habitats (ND = Neritic developmental, NI = Neritic inter-nesting, OI = Oceanic inter-nesting, M= Migratory, NF = Neritic feeding, OD = Oceanic developmental) and age classes.

Size class	Habitat	Mean	Maximum	References
Post hatchling	OD	2.5	9.3	Salmon *et al.* (2004)
Juvenile	ND	2.1	28	Blumenthal *et al.* (2010)
Juvenile	ND	2.1	2.9	Francke and Hargrove (2013)
Juvenile	ND	21.7	37.1	Hays *et al.* (2007)
Juvenile	ND	3.2	6.1	Makowski *et al.* (2006)
Juvenile	ND	2.9	19.5	Southwood *et al.* (2003)
Juvenile	OD		91	Swimmer *et al.* (2006)
Juvenile + Adult	ND + NF		7.9	Hazel *et al.* (2009)
Juvenile + Adult	ND + NF	28.4	54.5	Seminoff *et al.* (2006)
Adult	NI + OI	17.1	65.2	Cheng *et al.* (2013)
Adult	NI	2.8	13.1	Fuller *et al.* (2009)
Adult	NI	4.6		Glen *et al.* (2001)
Adult	NI	13.2	32	Hays *et al.* (2000)
Adult	NI	15.2		Hays *et al.* (2004b)
Adult	NI	4.4-7.8	24.8	Hochscheid *et al.* (1999)
Adult	NI	20.5	59	Yasuda and Arai (2009)
Adult	M	28.7		Hays *et al.* (2001)
Adult	M	38.8	138	Rice and Balazs (2008)

Hawksbill turtles (*Eretmochelys imbricata*)

Hawksbill turtles in the SWIO nest primarily at the Seychelles Archipelago (Allen *et al.*, 2010, Mortimer, 1984, Mortimer *et al.*, 2011a) (Fig. 2.5), with the annual number of nesting females in the region estimated to exceed 2500 (Appendix A). The long-term population trend in the SWIO has shown a decrease in nesting numbers, with most populations declining or being remnants of once large populations (Mortimer & Donnelly, 2007). Declines over the long term are evident for several populations, including populations in Kenya (Frazier, 1982, Okemwa *et al.*, 2004a), Seychelles (Mortimer, 2004, Mortimer, 1984, Mortimer, 2006), Mozambique (Mortimer & Donnelly, 2007), and Tanzania (Frazier, 1982, Howell & Mbindo, 1996). However recent increases have been seen at the Seychellois Islands of Cousin, Cousine (Allen *et al.*, 2010), and D'arros (Mortimer *et al.*, 2011a), and the French island Juan de Nova (Lauret-Stepler *et al.*, 2010). Genetic studies on the populations in the region are limited and in the absence of evidence to the contrary, this population will be treated as one stock. Satellite-tracking studies of post-nesting hawksbill females (Fig. 2.6) show that they tend to remain close to their rookeries, with few individuals travelling great distances between nesting beaches and foraging areas (Hawkes *et al.*, 2012, Horrocks *et al.*, 2011, Moncada *et al.*, 2012, Mortimer & Balazs, 2000, Musick & Limpus, 1997, Van Dam *et al.*, 2008).

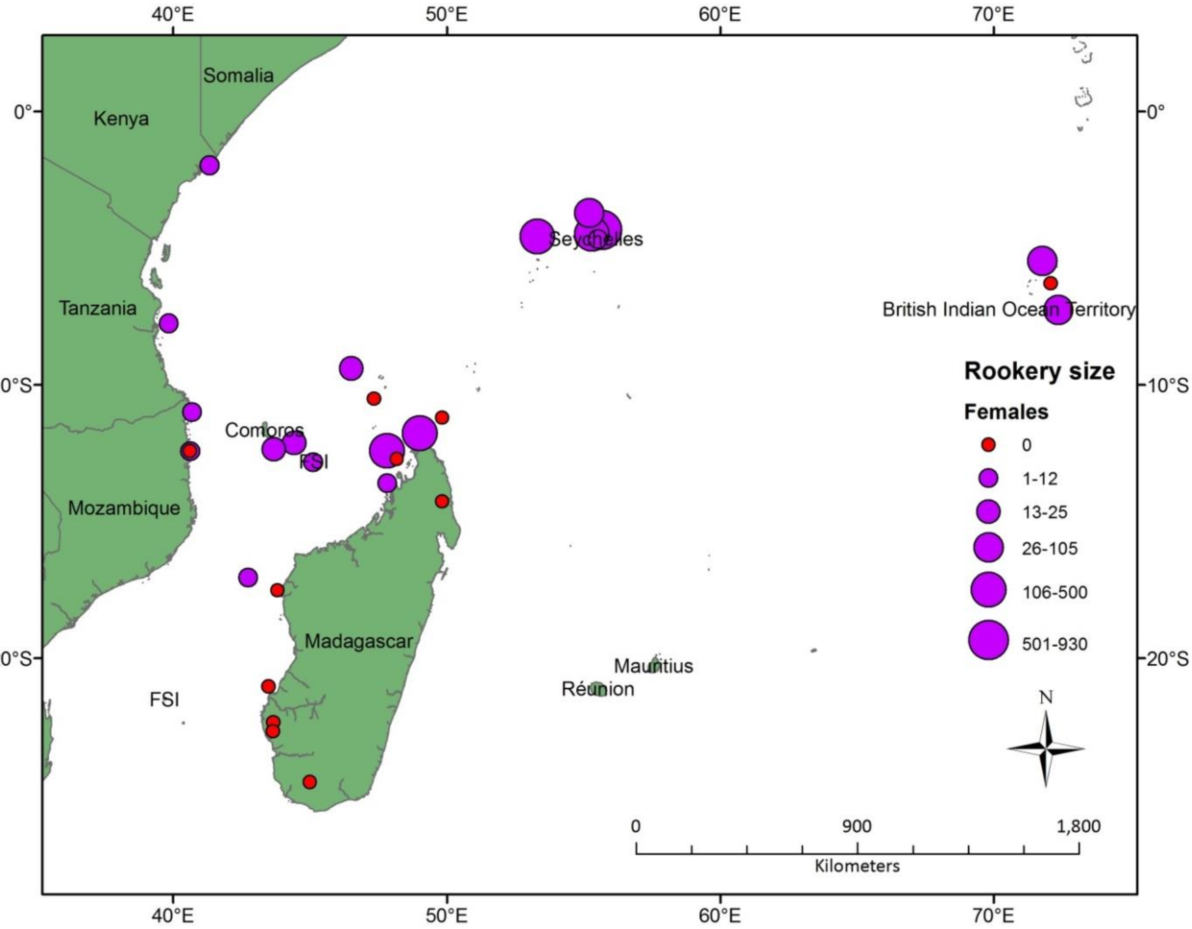


Figure 2.5 Distribution of hawksbill nesting sites (rookeries) in the SWIO as per the OBIS/SEAMAP online database (Kot *et al.*, 2013), supplemented with additional published literature on nesting areas (Appendix A). Rookery sizes are reported as number of nesting females per annum. Rookeries with 'zero' females (shown in red) are rookeries where nesting has been confirmed but the rookery sizes are currently unquantified.

Sexual maturity in hawksbills at populations outside the SWIO is reached at 17 - 40 years (Bell & Pike, 2012, Limpus *et al.*, 2008, Snover *et al.*, 2013). Although no data are available for rookeries in the region, age to maturity is estimated at >30 years (Mortimer & Donnelly, 2007). Reproductive longevity of hawksbills nesting at Cousin Island ranged from 17 – 20 years (Mortimer & Bresson, 1999). Therefore, the maximum reproductive age is estimated at more than 50 years, and generation length at approximately 40 years. Hawksbill turtles show remigration intervals of 2 – 3 years at Cousin Island (Mortimer & Bresson, 1999), with a female laying a mean of 3.6 – 4.0 clutches per nesting season (Diamond, 1976, Mortimer & Bresson, 1999), comprising an average of 177 – 182 eggs per clutch (Diamond, 1976, Hitchins *et al.*, 2004b).

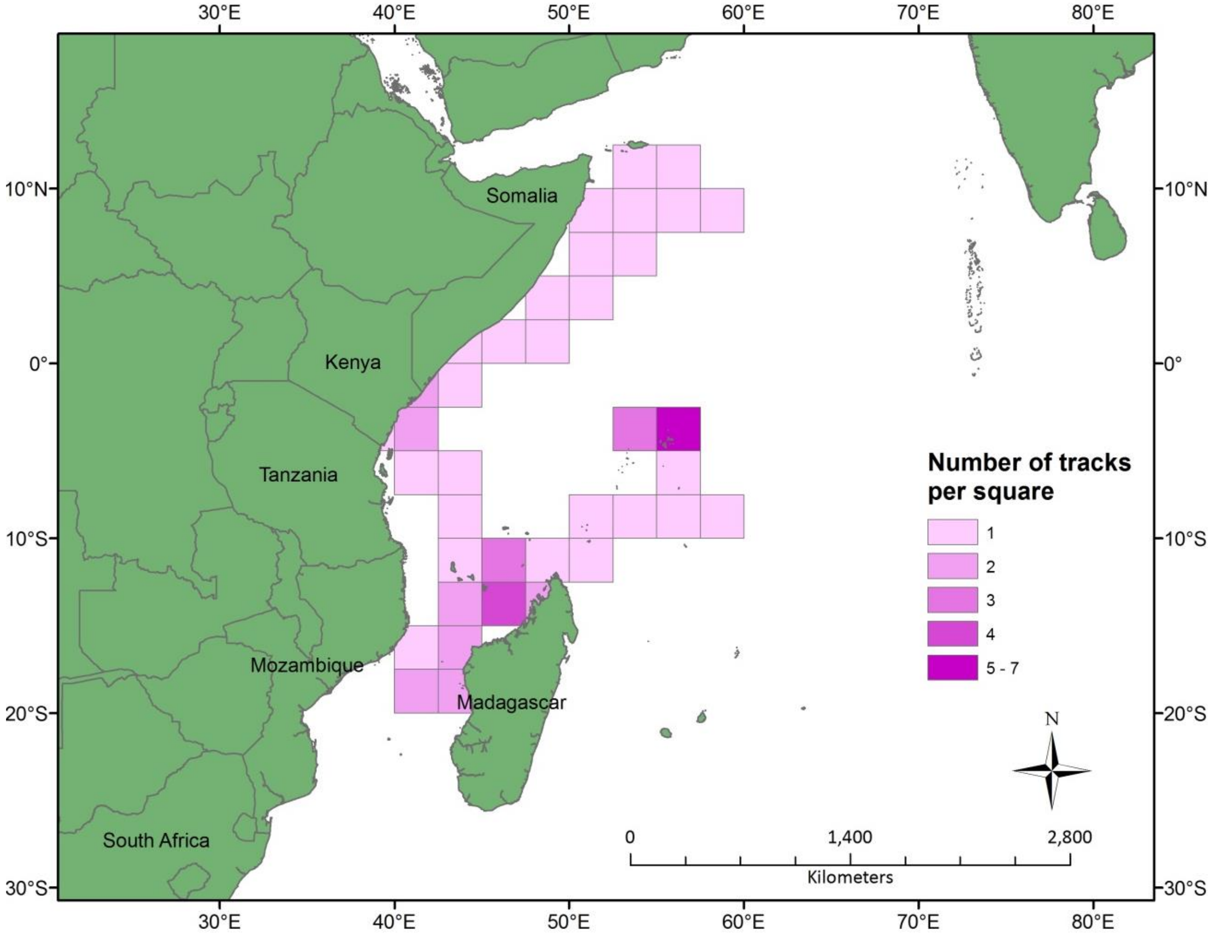


Figure 2.6 Approximate distribution of mostly post-nesting hawksbill turtles in the SWIO generated from published and unpublished satellite tracking studies (Appendix B). Densities are shown as number of tracks per 2.5° x 2.5° square.

Much like green turtles, young hawksbills spend their first few years in the pelagic environment in association with sargassum rafts (Carr, 1987b), exhibiting an omnivorous diet (Bjorndal, 1997). After recruiting to neritic feeding habitats they mostly forage over coral reefs, rocky substrates, sea grass pastures and mangrove-fringed bays, (Berube *et al.*, 2012, Bjorndal & Bolten, 1988, Bjorndal, 1997, Blumenthal *et al.*, 2009a, León & Bjorndal, 2002, Rincon-Diaz *et al.*, 2011), targeting sponges and soft corals (Frazier, 1984, Frazier, 1985, Meylan, 1988, Obura *et al.*, 2010).

Diving depths of hawksbill turtles differ among habitats and age classes (Table 2.2). Only one study has been conducted on diving depths in the SWIO on juveniles in a neritic developmental habitat. The maximum-recorded diving depth for juvenile hawksbill turtles in neritic developmental grounds is 91 m with the deepest recorded depth for a population in the SWIO being 26 m. The average diving depth of juveniles is 5.1 – 10.4 m, with an average of 8.2 m for the SWIO population. The maximum diving depth for adults in neritic inter-nesting areas is 55.3 m with averages ranging between 5.7 – 11.3 m. In neritic feeding grounds, the maximum-recorded diving depth of adult hawksbill turtles is 48 m with an average of 19.2 m.

Table 2.2 Mean and maximum dive depths (in meters) of hawksbill turtles across various habitats (ND = Neritic developmental, NI = Neritic inter-nesting, NF = Neritic feeding) and age classes.

Size class	Habitat	Mean	Maximum	References
Juvenile	ND	8	91	Blumenthal *et al.* (2009b)
Juvenile	ND	6.9	9	Houghton *et al.* (2003)
Juvenile	ND	10.4		Okuyama *et al.* (2012)
Juvenile	ND	6.8	72.1	Van Dam and Diez (1996)
Juvenile	ND	8.2*	26*	von Brandis *et al.* (2010)
Juvenile	ND	5.1	23.3	Witt *et al.* (2010)
Adult	NI	5.7	20.6	Bell and Parmenter (2008)
Adult	NI	11.3	55.3	Houghton *et al.* (2008a)
Adult	NF	19.2	48	Storch *et al.* (2005)

*** Values for populations in the SWIO.**

Leatherbacks (*Dermochelys coriacea*)

Leatherbacks nest along the coast of southern Mozambique and northern KwaZulu-Natal in South Africa (Hughes, 1974b, Nel *et al.*, 2013a, Videira *et al.*, 2011, Fig. 2.7). The population is small, with the total number of nesting females in the region estimated to be <100 per annum (see Appendix A), and are believed to be from one genetic stock (Dutton *et al.*, 1999). In spite of strong investment in leatherback conservation in South Africa, the number of nesting females has remained low, but stable for several decades (Nel *et al.*, 2013a). Satellite-tagging studies have shown that female leatherbacks remain within 100 km of nesting beaches during inter-nesting, and undergo tremendous long-distance migrations between nesting seasons (Eckert *et al*., 2012, Fig. 2.8). These migrations span large oceanic areas comprising complex routes (Luschi *et al.*, 2003c), largely following the prevailing currents (Lambardi *et al.*, 2008, Luschi *et al.*, 2003a, Luschi *et al.*, 2003c). However, they are also capable of remaining in the Indian Ocean coastal waters, foraging and rarely moving beyond 100 km from shore (Robinson *et al.*, 2013).

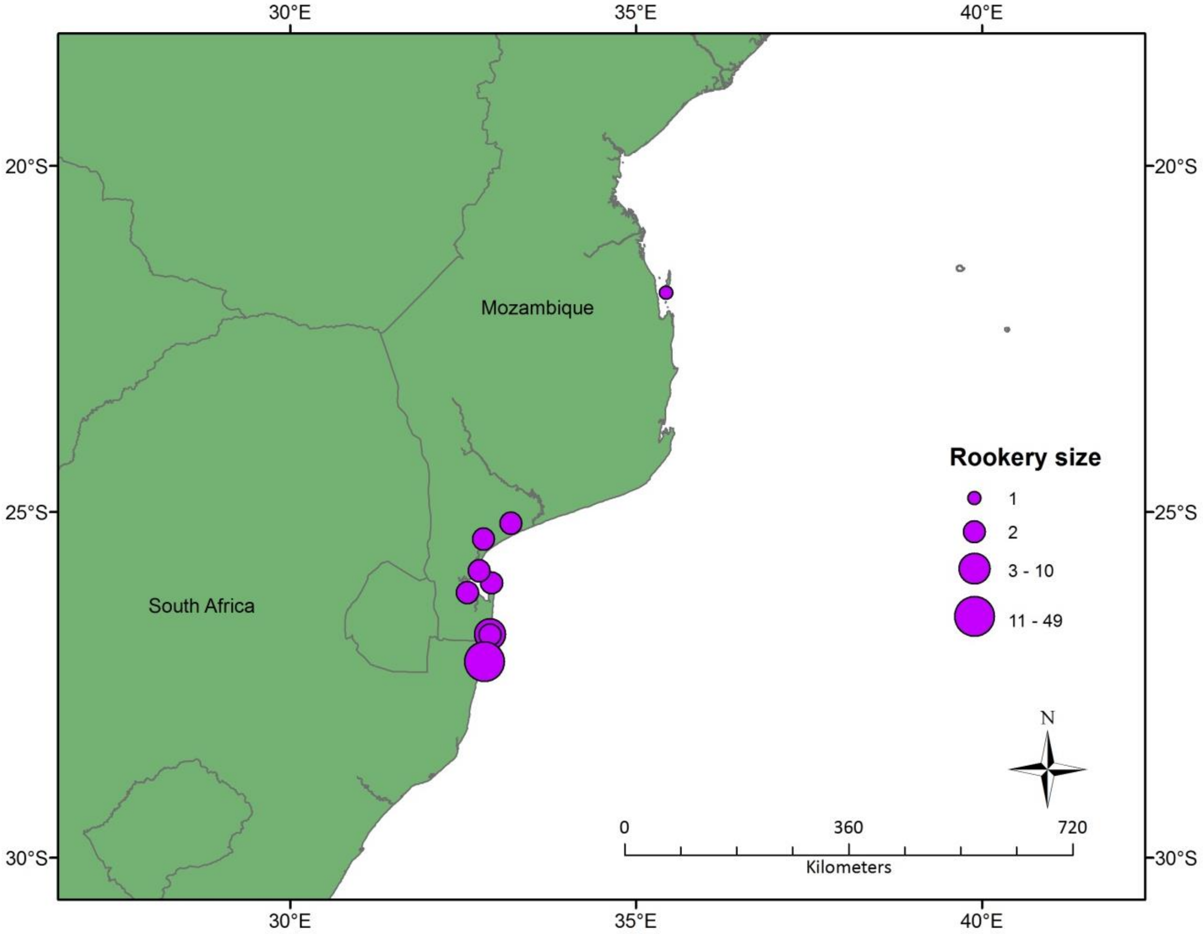


Figure 2.7 Distribution of leatherback nesting sites (rookeries) in the SWIO as per the OBIS/SEAMAP online database (Kot *et al.*, 2013), supplemented with additional published literature on nesting areas (Appendix A). Rookery sizes are reported as number of nesting females per annum.

Leatherbacks grow rapidly from hatchling to adults (Jones *et al.*, 2011) compared to other species, and age at sexual maturity for leatherback turtles is attained at 7-29 years (Avens *et al.*, 2009, Dutton *et al.*, 2005, Jones *et al.*, 2011, Zug & Parham, 1996). No studies have been conducted on the age to maturity for the population in the SWIO and it is thus estimated at

16 years (range 13.1 – 20.2) based on growth curve calculations (Jones *et al.*, 2011). With reproductive longevity in the South African population varying between 16 to 19 years (Nel *et al.*, 2013a), the maximum reproductive age for these turtles is estimated at 32 – 35 years, and generation length at approximately 24.0 – 25.5 years. An average period of 2.9 ± 1.8 years lapses between successive nesting seasons of leatherback turtles in the SWIO (Nel *et al.*, 2013a). A female deposits an average of 6.7 ± 1.5 clutches in each nesting season (Nel *et al.*, 2013a), with each clutch comprising a mean number of 84.1 ± 2.6 (SE) eggs (De Wet, 2012)

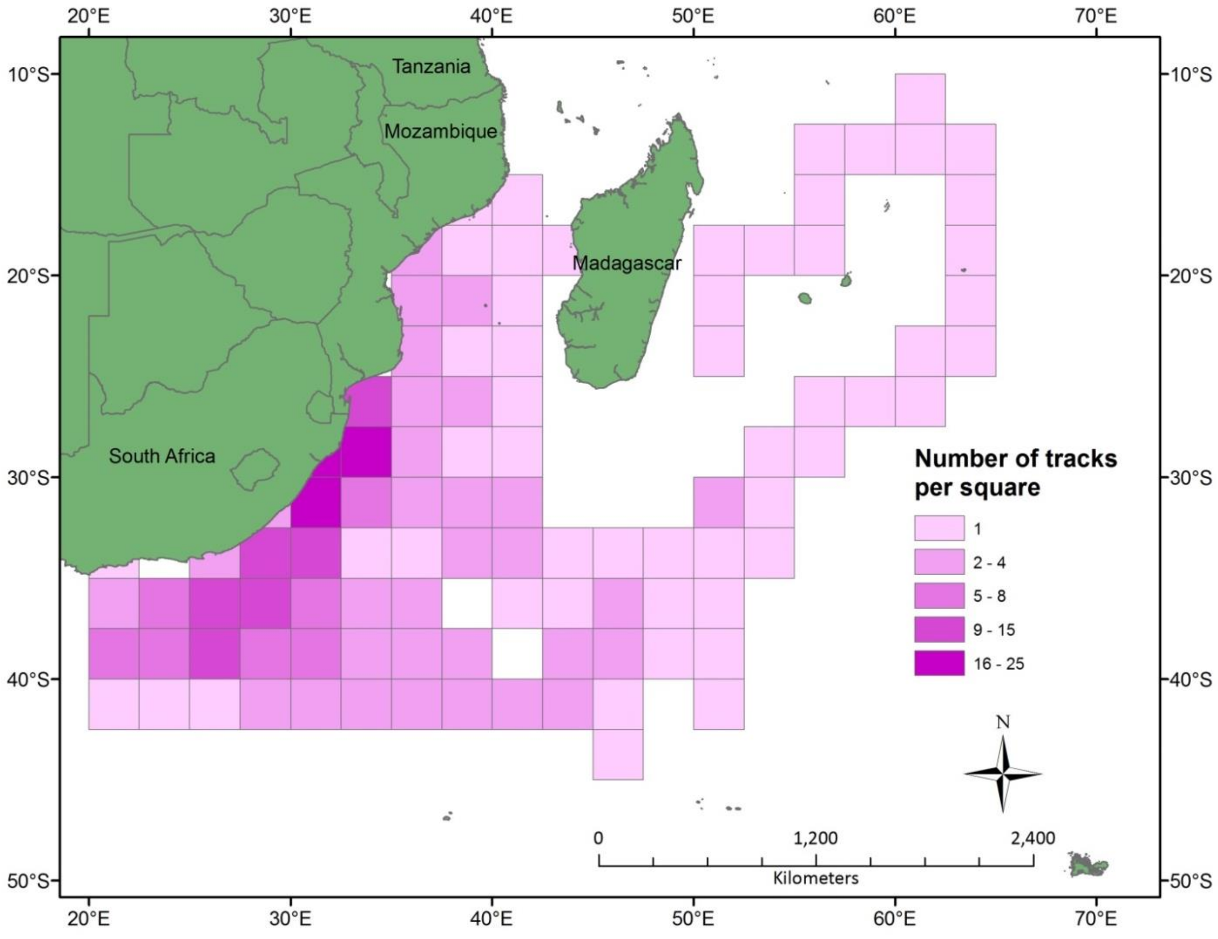


Figure 2.8 Approximate distribution of (mostly) post-nesting leatherback turtles in the SWIO generated from published and unpublished satellite tracking studies (Appendix B). Densities are shown as the number of tracks per 2.5° x 2.5° square, with larger densities given in darker shades of purple.

The diet of leatherback turtles consists primarily of gelatinous zooplankton throughout their lives (Bjorndal, 1997, Dodge *et al.*, 2011, Fossette *et al.*, 2011, Heaslip *et al.*, 2012, Robinson, 2014, Salmon *et al.*, 2004). Leatherback turtles have long been thought to be pelagic for most of their lives, however they can forage close to shore and over continental shelves (Benson *et al.*, 2011, Hoffman & Fritts, 1982, James *et al.*, 2006b, Lee & Palmer, 1981, Robinson, 2014). The distribution of this species seems to be highly dependent upon the distribution of the prey items that they consume (Lazell Jr, 1980, Leary, 1957), which often aggregate near convergent zones and water-mass boundaries (Collard, 1990). These turtles also feed throughout the water column to great depths often feeding at the deep-scattering layer at night (Eckert *et al.*, 1989b).

Diving depth of leatherbacks vary among age classes and habitats (Table 2.3). The maximum-recorded diving depth for post hatchlings is 17.1 m with an average of 5.3 m. Adults in neritic inter-nesting habitats dive to a maximum depth of 146 m with averages ranging from 9.4 – 33.1 m. Maximum adult diving depth, spanning across both oceanic and neritic inter-nesting habitats, is 1300 m, averaging 45.4 – 151.8 m. During migration in adults, the maximum-recorded diving depth is 640 m with an average of 51.7 m; however, these turtles can dive to a maximum depth of 1250 m during feeding, with averages ranging from 52.9 – 70.5 m.

Table 2.3 Mean and maximum dive depths (in meters) of leatherback turtles across various habitats (ND = Neritic developmental, NI = Neritic Inter-nesting, NF = Neritic Feeding, OI = Oceanic Inter-nesting, OF = Oceanic Feeding, M = Migratory, OD = Oceanic Developmental) and age classes.

Size class	Habitat	Mean	Maximum	References
Post hatchling	OD	5.3	17.1	Salmon *et al.* (2004)
Adult	NI	33.1	62	Eckert *et al.* (1996)
Adult	NI	9.4	83.8	Fossette *et al.* (2007)
Adult	NI	11.8		Fossette *et al.* (2008a)
Adult	NI	16	64	Reina *et al.* (2005)
Adult	NI	19.1	124	Southwood *et al.* (1999)
Adult	NI	33	146	Southwood *et al.* (2005)
Adult	NI	14.6	35.2	Wallace *et al.* (2005)
Adult	NI + OI	103.2	475	Eckert *et al.* (1986)
Adult	NI + OI	61.6	1300	Eckert *et al.* (1989b)
Adult	NI + OI	45.4	56.8	Eguchi *et al.* (2006b)
Adult	NI + OI	151.8	462.4	Fossette *et al.* (2010)
Adult	M	51.7	640	Hays *et al.* (2004a)
Adult	OF	52.9	1250	Houghton *et al.* (2008b)
Adult	OF	70.5	940	Sale *et al.* (2006)
Adult	NF + OF		96	James *et al.* (2006a)

Loggerhead turtles (*Caretta caretta*)

Loggerhead turtles in the SWIO occur throughout the temperate and tropical regions (Fig. 2.9), with nesting mainly occurring along the beaches of southern Mozambique and northern parts of South Africa (Hughes, 1974c, Nel *et al.*, 2013a, Videira *et al.*, 2011), with a small nesting population occurring in Madagascar (Rakotonirina & Cooke, 1994). The total estimated number of annual nesting females in the region is >590 (Appendix A). The population in South Africa (the biggest rookery in the region) has shown an increase in population over several decades (Nel *et al.*, 2013a). The rookeries in the SWIO are believed to be from one genetic stock (Shamblin *et al.*, 2014). The population in the SWIO shows a mixed migration strategy with some moving northward to feeding grounds in Mozambique, Tanzania, and Kenya, some moving eastward migrating towards Madagascar and some southward to Cape Agulhas and the Atlantic Ocean (Luschi *et al.*, 2006, Papi *et al.*, 1997) (Fig. 2.10).

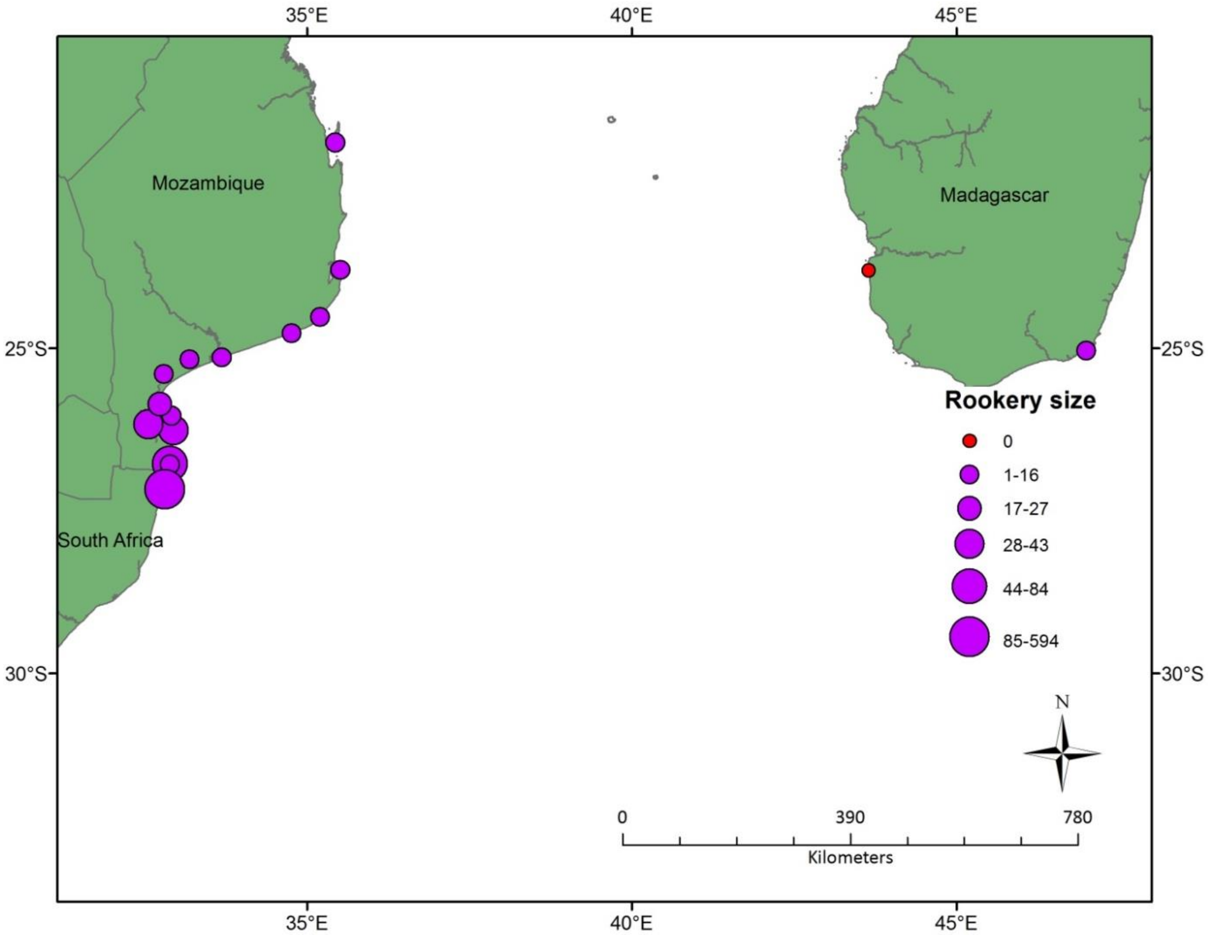


Figure 2.9 Distribution of loggerhead nesting sites (rookeries) in the SWIO as per the OBIS/SEAMAP online database (Kot *et al.*, 2013), supplemented with additional published literature on nesting areas (Appendix A). Rookery sizes are reported as number of nesting females per annum. Rookeries with 'zero' females (shown in red) are rookeries where nesting has been confirmed but the rookery sizes are currently unquantified.

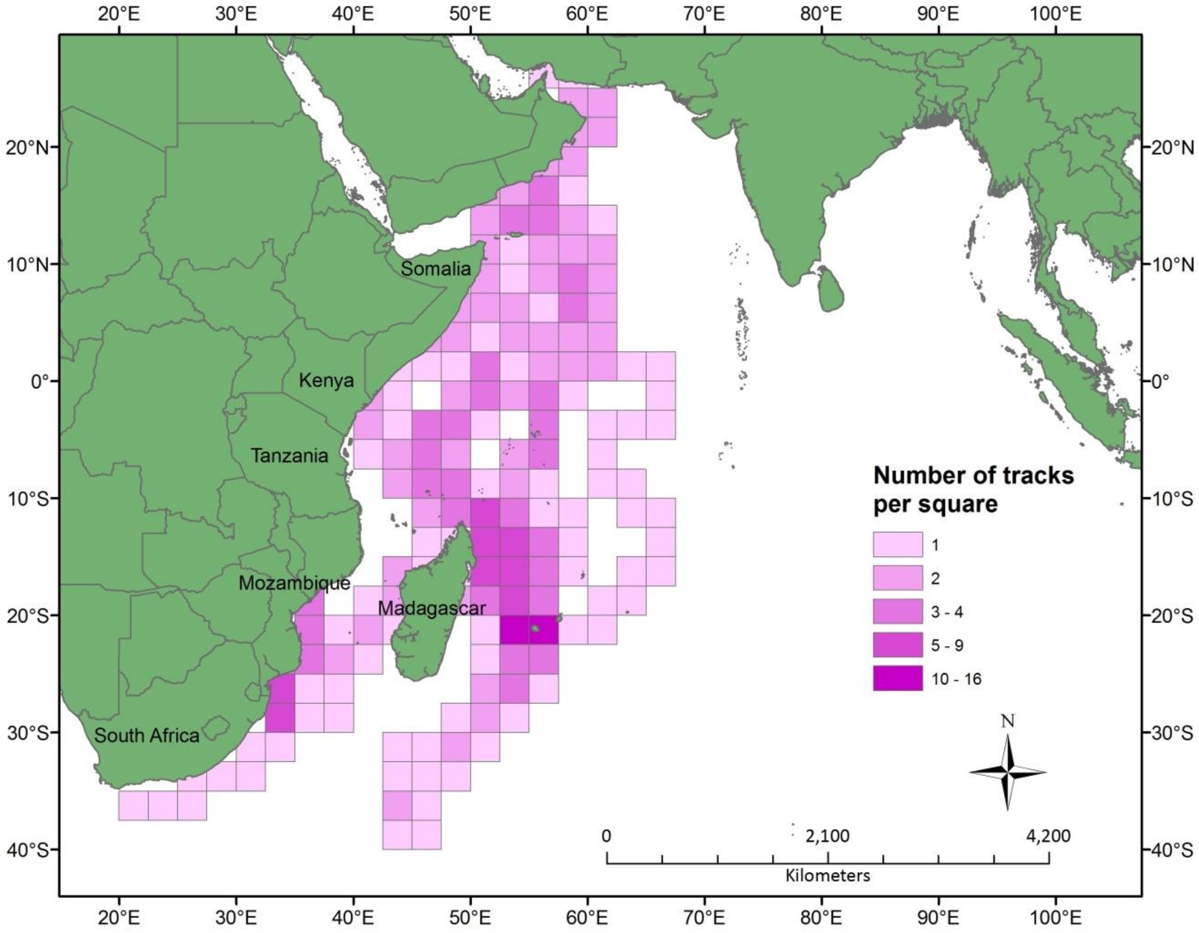


Figure 2.10 Approximate distribution of mostly post-nesting and bycaught juvenile/sub-adult loggerhead turtles in the WIO generated from published and unpublished satellite tracking studies (Appendix B). Densities are shown as number of tracks per 2.5° x 2.5° square.

The mean age to sexual maturity for loggerheads in the SWIO has been estimated at 36.2 ± 7.71 (SD) years (Tucek *et al.*, 2014), with the maximum observed reproductive longevity being 18 years (Nel *et al.*, 2013a). The maximum reproductive age of loggerhead turtles are thus calculated as 54 years with the generation length being 45 years. Loggerhead turtles in the SWIO have a remigration interval of 3.0 ± 2.2 years (Hughes, 1974a, Nel *et al.*, 2013a), laying 3.7 ± 0.8 clutches per season (Hughes, 1974c, Nel *et al.*, 2013a, Rees *et al.*, 2010), with each of the clutches having 106.6 ± 0.9 (SE) eggs (De Wet, 2012).

After hatching, loggerhead turtles forage in the pelagic environment for the first few years of life in association with sargassum floats (Carr, 1987b, Hughes, 1974c) exhibiting a carnivorous diet (Bjorndal, 1997, Boyle & Limpus, 2008, Hughes, 1974c). When recruiting to the neritic habitat this species undergoes a gradual shift in diet by still feeding on epipelagic organism while gradually increasing the intake of benthic organisms (Bolten, 2003, Casale *et al.*, 2008, Parker *et al.*, 2005, Peckham *et al.*, 2011). At neritic feeding habitats, this species forage over both hard and soft-bottomed habitats. They are also thought to be opportunistic feeders because of the wide range of species consumed and environments used (Thomson *et al.*, 2012). Loggerhead turtles from the population in South Africa feed predominantly in coastal habitats in the Mozambique Channel as seen from tag recoveries (Luschi *et al.*, 2003b), satellite telemetry (Luschi *et al.*, 2003b, Papi *et al.*, 1997) and stable isotope analysis (Robinson, 2014).

Diving depths of loggerhead turtles vary among age classes and habitats occupied (Table 2.4). The maximum diving depth recorded for loggerhead turtles is 233 m in oceanic inter-nesting habitat with an average diving depth of 54 m. Juveniles and sub-adults in the oceanic habitat dive to a maximum of 178 m, whereas adults in the oceanic feeding and neritic feeding areas dive to a maximum of 174.5 m. In the neritic inter-nesting areas adult loggerhead turtles dive to a maximum of 120 m with the average diving depth between studies ranging from 6.1 – 8.7 m.

Table 2.4 Mean and maximum dive depths (in meters) of loggerhead turtles across various habitats (ND = Neritic developmental, NI = Neritic Inter-nesting, NF = Neritic Feeding, OI = Oceanic inter-nesting, OF = Oceanic Feeding, M = Migratory, OD = Oceanic Developmental) and age classes.

Size class	Habitat	Mean	Maximum	References
Juvenile + sub-adult	OD		178	Polovina *et al.* (2003b)
Adult	NF + OF		150	Hatase *et al.* (2007)
Adult	OF		174.5	Hochscheid *et al.* (2005)
Adult	NI	8.7	70	Houghton *et al.* (2002)
Adult	NI	5.2	120	Minamikawa *et al.* (1997)
Adult	NI	6.1		Minamikawa *et al.* (2000)
Adult	OI NI	39		Sakamoto *et al.* (1993)
Adult	OI	54	233	Sakamoto *et al.* (1990a), Sakamoto *et al.* (1990b)

Olive ridleys (*Lepidochelys olivacea*)

Olive ridley turtles are widely distributed in the South Western Indian Ocean although they are mostly vagrants (Bourjea, 2012). Resultantly, data on nesting locations and rookery size, genetics distinctiveness, behaviour and life history characteristics are limited. It is estimated that there are > 1000 annual nesting females within WIO-RMU (Appendix A), with two distinct nesting behaviours i.e. conventional nesting and synchronised mass nesting or *arribada* (which is unique to this species). A few records of nesting do exist for the East African coast from South Africa, Madagascar (Frazier 1975), Mozambique and Kenya (Kot *et al.*, 2013) (Fig. 2.11). This infrequent nesting combines to less than 10 females nesting annually within the SWIO region. Due to the data paucity, even at sea distribution is poorly established and hence the RMU boundaries span the entire WIO region with nesting predominantly in Oman and India. Very little published or online satellite tracking information is available for olive ridley turtles in the SWIO (Fig. 2.12) with most information available for tracking of post-nesting females from the dominant rookeries.

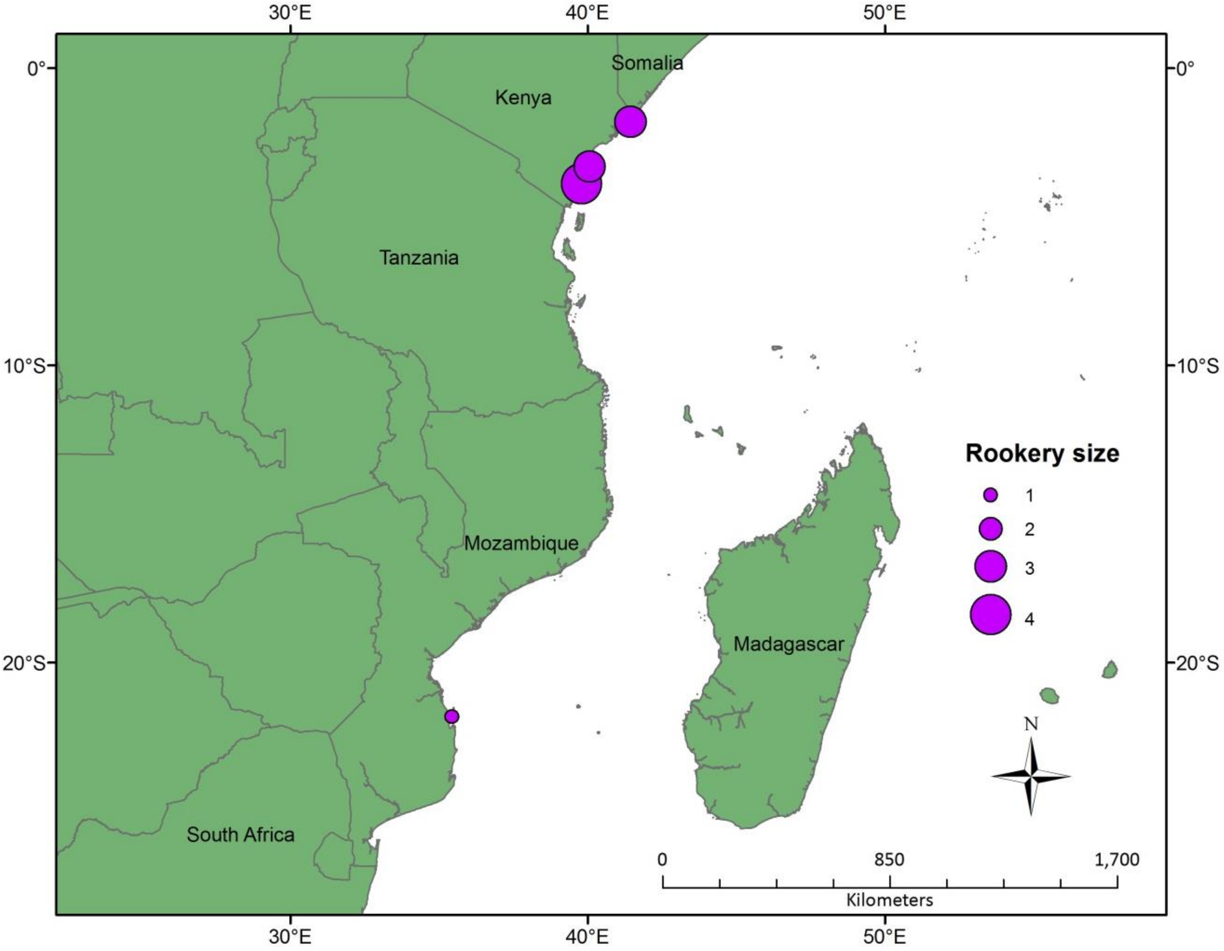


Figure 2.11 Distribution of olive ridley nesting sites (rookeries) in the SWIO as per the OBIS/SEAMAP online database (Kot *et al.*, 2013), supplemented with additional published literature on nesting areas (Appendix A). Rookery sizes are reported as number of nesting females per annum.

The age to sexual maturity for this species is inferred from studies outside the region and estimated at 13 years (Zug *et al.*, 2006). However, following the trends from other hard-shelled turtles this seems an underestimate. There is a lack of data available on the feeding and habitat use of this species but most research reports a omnivorous diet throughout their lives with plasticity in the use of neritic and oceanic habitats (Bolten, 2003).

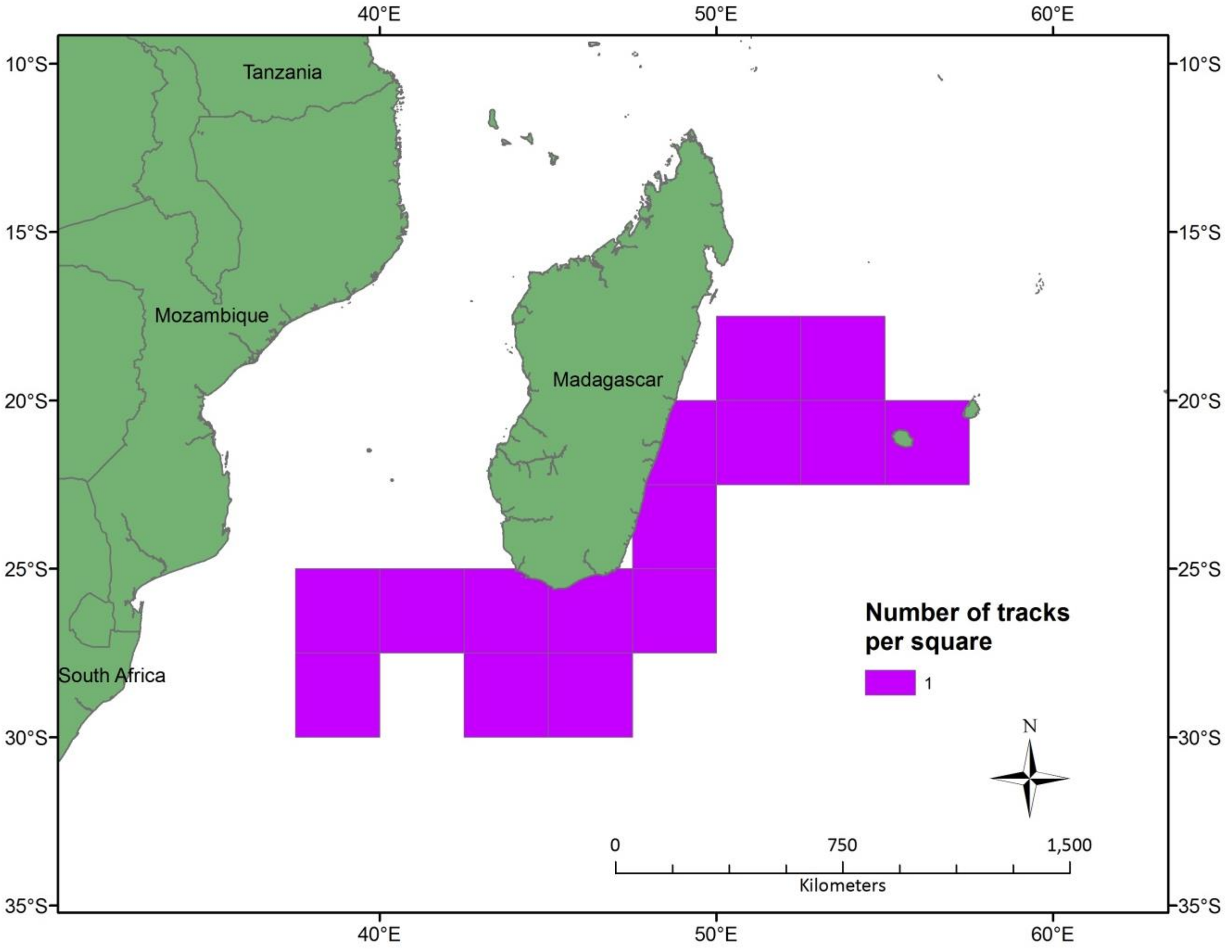


Figure 2.12 Approximate distribution of the only satellite tracked olive ridley turtle in the SWIO generated from satellite tracking studies (as per Appendix B). Densities are shown as number of tracks per 2.5° x 2.5° square.

The diving depths of olive ridley turtles vary among age classes and habitats occupied (Table 2.5). The maximum recorded diving depth for olive ridley turtles are for juveniles in the oceanic developmental habitat with a depth of 408 m, whereas the maximum-recorded depths of olive ridley turtles in the neritic and oceanic feeding habitats is 200 m with an average diving depth of 39.8 m.

Table 2.5 Mean and maximum dive depths (in meters) of olive ridley turtles across various habitats (NF = Neritic Feeding, OF = Oceanic Feeding, OD = Oceanic Developmental) and age classes.

Size class	Habitat	Mean	Maximum	References
Adult	NF + OF	39.8	200	(McMahon *et al.*, 2007)
Juvenile + sub-adult	OD		254	(Polovina *et al.*, 2003a)
Juvenile	OD		408	(Swimmer *et al.*, 2006)

Sea turtles and fisheries

Sea turtles are migratory species with transoceanic movements (Bourjea *et al.*, 2013, Luschi *et al.*, 2006) that bring them into contact with fisheries across their geographic distributions, and hence fisheries have been implicated as a major reason for the decline in sea turtle populations around the globe (Lewison *et al.*, 2004a, Wallace *et al.*, 2011a, Wallace *et al.*, 2011b, Wallace *et al.*, 2013b). In the SWIO, the fisheries that are most likely to interact with sea turtles include open ocean fisheries like longlines, and purse seines, and coastal fisheries such as trawl, gillnet, and beach seine fisheries. Open ocean fisheries frequently interact with the pelagic juvenile stages of sea turtles (Amandé *et al.*, 2008, Anderson, 2009), whereas coastal fisheries interact with sea turtles in either feeding and inter-nesting habitats where they occur in high densities. Any of these fisheries activities that impact on larger age classes with higher reproductive value (the relative contribution of an size/age-class to reproduction, Fisher, 1930) will have a greater impact on the population (Wallace *et al.*, 2008). However, the protection of large age classes in the population is imperative to the survival of the species (Crouse *et al.*, 1987, Heppell, 1998, Heppell & Crowder, 1998). For this dissertation, the bycatch impact of five fisheries will be quantified and the relative risk of the fisheries to turtles compared. The fisheries included in the analyses are both the large scale offshore industrial fisheries (including longline and purse seine fisheries) and the smaller scale coastal fisheries that are often artisanal (including prawn trawl, gillnet, and beach seine fisheries).

Fisheries of the SWIO

There is great ichthyofaunal richness in the SWIO due to the variety of habitats and oceanographic conditions that supports a variety of fisheries (Van der Elst, 2012). These fisheries are of great social and economic importance in the region, providing both income and protein to the populace of the region (Van der Elst, 2012). Great contributions to the national economies of the region are made by fisheries, with smaller island nations often being more dependent upon these activities as a source of foreign exchange than mainland nations (Kimani *et al.*, 2009, Van der Elst, 2012, van der Elst *et al.*, 2005).

The fisheries in the region range from industrial operations to artisanal fisheries (van der Elst *et al.*, 2005). The industrial fisheries require high capital and technological investment to enable them to operate over long distances for extended periods. They operate in both the economic exclusion zones (EEZ) of countries and the high seas. In contrast, artisanal fisheries require low capital investment and operate over short distances, mostly along the coast, due to the limitations of boats and equipment. These fisheries are concentrated in the coastal areas and in near shore reefs and continental drop-offs.

The gear types used in the different sectors differ and are mainly dependent upon the species targeted by the fisheries. The industrial fisheries use mainly longline and purse seine gear for targeting tuna and tuna-like species, and trawl gear for prawns, langoustines, lobster, and crabs (ASCLME/SWIOFP, 2012, WIOFISH, 2011). Artisanal fishers by comparison employ a variety of fishing gears, including gillnets, beach seine nets, hand lines and spears (WIOFISH, 2011). These fisheries operations target inshore demersal and small pelagic fish species, sea cucumbers, lobsters, crabs, prawns, bivalves and octopus (ASCLME/SWIOFP, 2012). Artisanal fishers often use a combination of gear types to maximise catches. Despite the lack of technological advancement in this sector, there has been a substantial increase in fishing effort in the past decade. This is a result of growing coastal populations and the increased demand for fish and fish products (ASCLME/SWIOFP, 2012), with landings from the artisanal sector surpassing that of the industrial sector (Kimani *et al.*, 2009). It is however necessary to have an understanding of each of the techniques used in fisheries, and their method of interaction with sea turtles, before the impact on sea turtles can be evaluated.

Longlines

Longline fishing operations target tuna, swordfish (*Thunnus* spp., *Katsuwonus* spp., *Auxis* spp., *Scomberomorus* spp., *Ziphiass* spp., *Makaira* spp., *Tetrapturus* spp., and *Istiophorus* spp.) and a variety of shark species (IOTC, 2013a). The baited hooks used to catch fish are attached to branch lines that are in turn attached to a mainline. The mainline might be several kilometres in length (Fig. 2.13; Beverly *et al.*, 2003, FAO, 2001-2014c). The lines are either set as drifting or anchored lines and left for hours at sea to capture the target species. Deep-set lines targeting tuna are often set during daylight hours at depths between 30 – 400 m while shallow-sets targeting swordfish are mainly set at night at depths between 15 – 100 m (Beverly *et al.*, 2003, FAO, 2001-2014c).

Globally longline fishing gear is seen as one of the major threats to sea turtle populations, especially leatherback and loggerhead populations (Casale, 2008, Donoso & Dutton, 2010, Lewison *et al.*, 2004b, Petersen *et al.*, 2009, Wallace *et al.*, 2013b). Interaction between sea turtles and longline gear occur when sea turtles are hooked (either externally or internally), or entangled in gear (Garrison & Stokes, 2012, Ulloa Ramirez & Gonzalez Ania, 1998, Watson *et al.*, 2005b). Location of sets have an effect on catch rates of sea turtles, with sets near thermal fronts or eddies showing higher catch rates. These highly productive areas are sought out by fishing vessels and sea turtles presumably follow these as well, increasing their likelihood of interaction with gear (Ferraroli *et al.*, 2004, Hays *et al.*, 2004a, Polovina *et al.*, 2004, Polovina *et al.*, 2000).

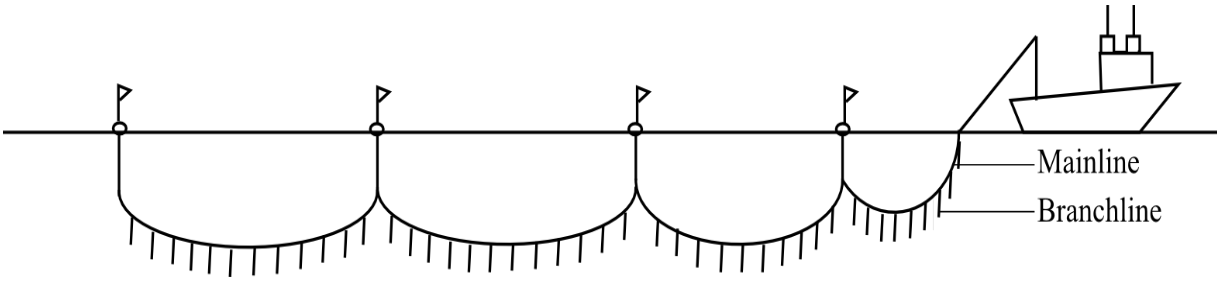


Figure 2.13 Schematic representation of longline gear (after Beverly *et al.* (2003).

The gear configuration affects the capture rates of sea turtles; shallow gear set at night to target swordfish have higher sea turtle capture rates than deeper set gear deployed during the day targeting tuna (Crowder & Myers, 2001, Gilman *et al.*, 2007, Ito & Machado, 2001, Petersen *et al.*, 2009), because sea turtles spend most of their time near the surface and mainly forage at night (Eckert *et al.*, 1989a, Parker *et al.*, 2005, Polovina *et al.*, 2003a). Furthermore, the mortality rate is much higher for deeper sets than for shallow sets (Caminas, 2004), due to the higher possibility of these sea turtles drowning before being able to surface (Kleiber & Boggs, 2000). Even in deep-set gear with the expectation of lower bycatch, 30 % of the hooks are still in the top 100 m of the water column and these hooks show more frequent interaction with sea turtles than those at deeper depths (Beverly & Robinson, 2004). Leatherback turtles dive deeper at night to access the deep zooplankton scattering layer (Eckert *et al.*, 1989b, Hays *et al.*, 2004a, Sale *et al.*, 2006) where targeted swordfish also forage (Brill & Lutcavage, 2001). Thus, greater interactions between leatherbacks and swordfish fisheries are to be expected. Further, light sticks used to attract swordfish may also attract leatherback turtles to longline gear, and they are indeed commonly observed to interact with sets where light sticks are used (Witzell, 1999).

Olfactory stimuli likely attract the carnivorous species such as loggerheads and olive ridleys to bait (Piovano *et al.*, 2004), even though fish are not a large part of their diet. However these two species may perceive the bait as easy prey increasing their interaction potential with gear (Parker *et al.*, 2005) and they are commonly hooked in the mouth (Arauz, 2000, Oceanic Fisheries Programme, 2001). Green, hawksbill and leatherback turtles are not likely to be attracted to bait as they are herbivores, spongivores and gelatinovores (Bjorndal, 1997) respectively. Another possible reason for high entanglement of leatherbacks in longline gear is the fact that leatherbacks cannot swim backwards (Davenport, 1987), and thus when encountering float lines they struggle to avoid lines and hooks and become entangled. Olive ridleys also frequently inspect the floats and may become entangled in the gear (Largacha *et al.*, 2005), increasing their interactions with gear.

Mortality in sea turtles captured may occur prior to haul back of gear or after being released, with the proportion of sea turtles that die on the line prior to retrieval ranging from 4 – 27 % (Aguilar *et al.*, 1995, Caminas, 2004, McCracken, 2000). Mortality prior to haul back is determined by the type of interaction (hooking status and entanglement), the set depth, and whether mitigation measures are used. Sea turtles hooked internally are more likely to die than those hooked externally or entangled (Santos *et al.*, 2012). Hard shelled sea turtles such as loggerheads and olive ridleys are commonly hooked internally as they swallow the bait and thus die more frequently than leatherbacks that are entangled (Ulloa Ramirez & Gonzalez Ania, 1998, Watson *et al.*, 2005a, Witzell & Cramer, 1995), or foul hooked (Watson *et al.*, 2005b, Witzell, 1984).

It is expected that the use of mitigation measures and proper handling and release practices can reduce the mortality of sea turtles during capture (Kobayashi & Polovina, 2005). However, the post-release mortality of sea turtles, and the effect of mitigation measures on this mortality is poorly research and quantified. Mortality estimates in the literature ranged from 25 % to 75 % mortality with no clear patterns (Aguilar *et al.*, 1995, McCracken, 2000, NMFS, 2001a, Parker *et al.*, 2001, Swimmer *et al.*, 2002). It is therefore not applied in this study.

Purse seine

In the SWIO, there are two types of purse seine fisheries, one targeting small pelagic fish like scads, sardines and mackerel, and one targeting tuna and tuna-like species (Lucas *et al.*, 2009). The main tuna species targeted by the purse seine industry are skipjack (*Katsuwonus pelamis*) and yellowfin tuna (*Thunnus albacares*) (ASCLME/SWIOFP, 2012). A purse seine net is a net constructed with a float line at the top and a series of purse rings hanging at the bottom through which a purse line runs (Fig. 2.14). The purse line is used to purse the net (close the bottom of the net) after fish are surrounded by the net (FAO, 2001-2014e). The gear is set at the surface in both coastal and high seas waters, but the net depths varies, reaching depths of up to 300 m (Hall & Roman, 2013), although the most commonly targeted depth in the WIO region is 70 m (IOTC, 2010). In the purse seine fishery two types of sets are distinguished; sets on free-swimming schools (FSC), and sets around fish aggregation devices (FAD) or floating objects.

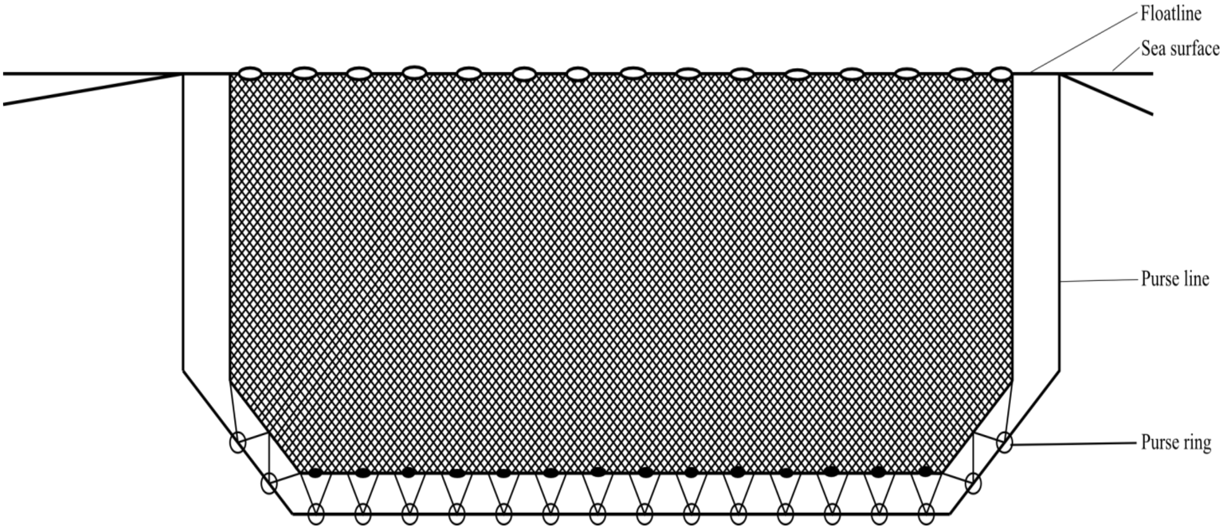


Figure 2.14 Schematic drawing of a purse seine net (after FAO (2001-2014e)).

Interactions between sea turtles and purse seine gear occur when the nets surround sea turtles together with the target catch. The captured sea turtles may be released alive, however some may drown when they become entangled in the mesh (Hall & Roman, 2013). Sea turtles are not able to keep up with a school of tuna therefore encounters between purse seiners with FSC and sea turtles are seen as chance encounters. As for the longline industry, purse seine vessels often seek out areas of high productivity to conduct fishing operations, however sea turtles are often also attracted to these areas of high productivity (Polovina *et al.*, 2001, Saba *et al.*, 2008).

Sets around FADs often have higher capture rates of sea turtles, because they may be attracted to floating objects in search of food and shelter (Witherington, 2002). Setting nets around floating objects started as an opportunistic method to increase catches of the target species (Le Gall *et al.*, 2000). However, fishers started deploying artificial FADs with rapid expansion of this practise during the 1990s, due to the efficiency of this method (Fonteneau *et* *al.*, 2000, Moreno *et al.*, 2007). Higher capture rates of sea turtles are associated with sets on FADs as turtles gather under or around FADs, or smaller post-hatchlings and juveniles crawl on top to rest (Amandè *et al.*, 2010, Fonteneau *et al.*, 2000). Sea turtles often become entangled in the netting that hangs below these FAD (Franco *et al.*, 2009) and these often become lost and result in ghost fishing (Jensen *et al.*, 2013, Wilcox *et al.*, 2014). Thus mitigation in purse seine fisheries now target FAD design, moving towards 'eco-friendly' FADs that do not have netting and are biodegradable (Franco *et al.*, 2012).

Prawn trawling

The target species of the prawn trawl fishing operations in the SWIO are *Penaueus indicus, Metapenaeus monoceros, P. monodon , P. japonicas, P. semisulcatus, P. latisulcatus,* with *P. indicus* and *M. monoceros* constituting 90 % of the landed prawn catch (ASCLME/SWIOFP, 2012). Prawn trawl fisheries use demersal otter trawls with up to four trawl sets on outrigger booms. The gear is made up of a cone-shaped net consisting of a body tapered into one or two cod-ends with lateral wings extending forward from the opening to funnel the catch into the net. The net is kept open by the two otter boards, attached to two lines that attach to the wings of the net. Most net designs have an extended top panel to prevent the catch from escaping over the top of the net (Fig. 2.15; (FAO, 2001-2014b).

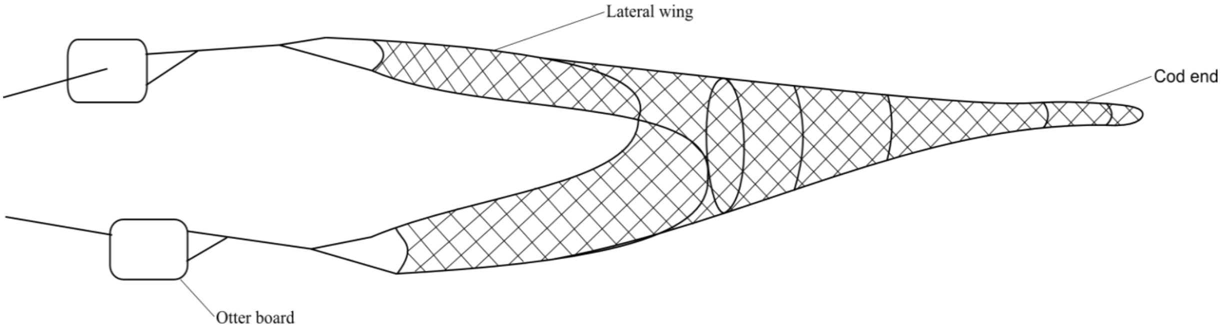


Figure 2.15 Schematic drawing of an otter trawl net (after FAO (2001-2014b).

Trawl nets are non-selective fishing gear that captures sea turtles together with the target species. These captured sea turtles may become comatose and eventually drown and die, or their physical condition may be weakened (Lutcavage & Lutz, 1997). The routine dive times of sea turtles are shorter than the average tow times of trawling vessels and the amount of time that a sea turtle can stay submerged may be further decreased during forced submergence due to increased physical activity to free themselves (Lutcavage & Lutz, 1997).

Factors that play a role in the tolerance to forced apnoea include size, activity and water temperatures, with larger sea turtles often being capable of longer voluntary dives than smaller sea turtles, making juveniles more vulnerable to forced submergence, especially in warmer months when the basal metabolic rate is higher (Lutcavage & Lutz, 1991).

Sea turtle mortality is related to tow times with an increase in mortality with an increase in tow times (Henwood & Stuntz, 1987, Kemmerer & Center, 1989, Watson & Seidel, 1980). Prior to drowning a sea turtle will enter a comatose state. When a sea turtle is brought on board a vessel in a comatose state, the time needed for recovery can be as much as 10 hours or even more (Lutz & Dunbar-Cooper, 1987). If a sea turtle is released back into the water in a comatose state they may drown because they cannot swim or surface to breathe (Kemmerer & Center, 1989). However, the selectivity of the trawl can be altered by adding various panels or grids into the design that function as bycatch reduction devices (BRDs) (Fennessy & Isaksen, 2007) or Turtle Excluder Devices (TEDs) which have proven to be quite successful (Brewer *et al.,* 2006, Brewer *et al.,* 1998).

Gillnets

A gillnet is a curtain of monofilament or multifilament mesh (of various sizes) in which fish are caught by getting gilled, entangled or enmeshed. Gillnets have a series of floats on the headline and weights on the foot line with the net spanned between them (FAO, 2001-2014d). The net can be set at the bottom, anchored but catching at the surface (like the bather protection nets) left to drift in the current (either at the surface or a specific distance below the surface) i.e. driftnets, or used to encircle fish (Fig. 2.16, FAO, 2001-2014d). Trammel nets are an alternative configuration to standard single layer gillnets that have three layers of netting (FAO, 2001-2014g). These nets are often used close to the ocean bottom as a stationary gear. Gillnet gears are highly variable within and among countries in the SWIO (Kiszka, 2012b), with elements such as set depth, mesh size, soak times and location of deployment being very diverse (Global, 2009, Kiszka, 2012b, Peckham *et al.*, 2007, WIOFISH, 2011). In general, however, gillnets are designed to be transparent, making them difficult for animals to detect in the water column (Rowe, 2007). These nets are also highly unselective with regards to target species, with selectivity depending on the mesh size of the net.

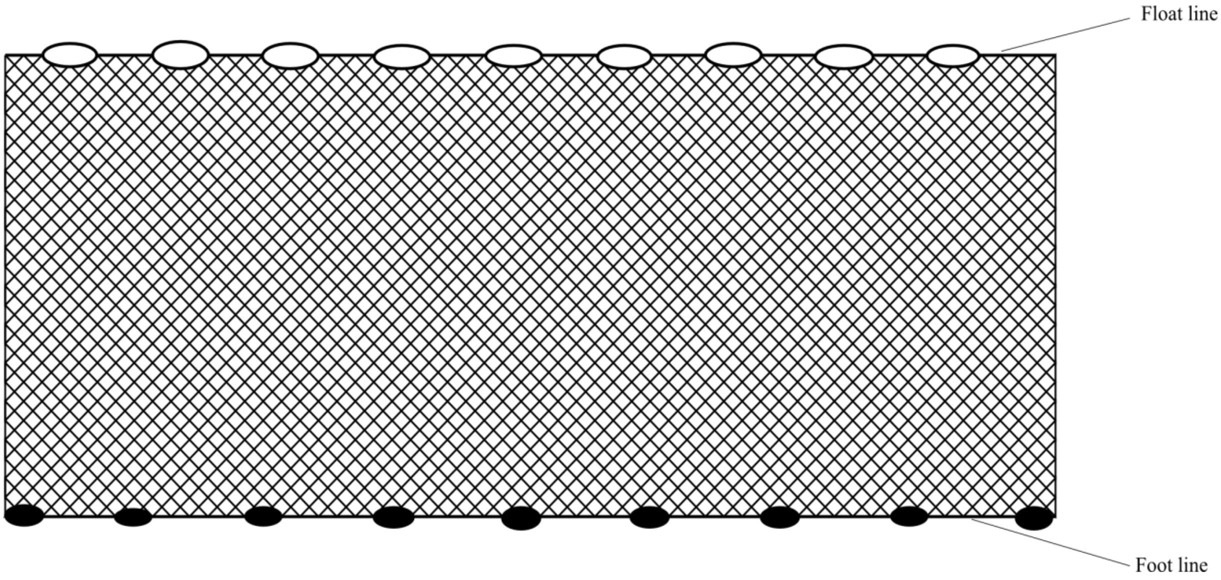


Figure 2.16 Schematic drawing of a gillnet (after FAO (2001-2014d)).

With the exception of the bather protection nets in KwaZulu-Natal (Brazier et al 2012), gillnet fisheries in the SWIO are mainly artisanal low-cost fishery targeting pelagic, demersal, and benthic species (FAO, 2001-2014f). Sea turtles become entangled in these nets and drown as they are not able to surface to breathe or serious injury may be inflicted to the sea turtle from lines and ropes that are used to support the gillnet in the water (Gilman *et al.*, 2010a). Sea turtles may break free from gear with a portion of the gear still attached that may cause further entanglement or injury.

Beach seine

Beach-seine gear is composed of a bunt (bag or loose netting) and long wings often lengthened with long ropes for towing the seine to the beach. The head rope with floats are on the surface and the foot rope is in constant contact with the bottom and the net therefore forms a barrier that prevents fish from escaping the enclosed net (FAO, 2001-2014a). Beach seines are often set from shore and hauled onto the beach, but can also be set some distance from shore but still in shallow water after which it is then hauled onto a boat. A seine net can be set from shore using a boat or by hand. Hauling can be done either by hand or motorized vehicle or tractor (Tietza *et al.*, 2011).

Within the SWIO this fishery is mainly artisanal with very little recording of data on the impacts of this fishery (WIOFISH, 2011). Sea turtles are captured in beach seine nets when these nets surround the sea turtle and the sea turtle is brought ashore with the target species. When the net surrounds the sea turtle, it is unlikely that the sea turtle will be able to escape. The mortality rate of sea turtles prior to being brought ashore is very low, as the sea turtles will still be able to surface to breathe for most of the time that it takes to haul the net (Tietza *et al.*, 2011), however between 8 – 42 % of captured sea turtles are killed for consumption, cultural and medicinal purposes (Kiszka, 2012a).

Fisheries management

One of the greatest challenges to fisheries management in the SWIO is the management of the "commons." The open nature of fisheries makes monitoring, controlling and surveying difficult, especially in remote regions. Fishers continually seek ways to increase landings and often choose less selective methods of fishing and gear types, especially in the artisanal fisheries, and see little incentive in complying with regulations (Kimani *et al.*, 2009). Fisheries regulations in most countries are out-dated and inappropriate, and enforcing these regulations is often inefficient due to a lack of capacity (Kimani *et al.*, 2009). It is also challenging fisheries managers in the region to formulate policies that maximize social and economic benefits for communities linked to the industry, and yet at the same time balance the sustainability of the fisheries resources. These challenges have led to a shift toward community-based and co-management approaches that allow local communities to have greater input in the formulation of regulations (Kimani *et al.*, 2009).

The increase in the global demand for fish has also led to an increase in illegal, unreported, and unregulated (IUU) fishing in the region in both the high seas and within EEZs (Kimani *et al.*, 2009). IUU fishing is possibly due to the lack of effective monitoring and control of activities. This undermines efforts to increase the sustainability of the industry and increase the impacts on bycatch species. This makes it essential to quantify the impact of fisheries on sea turtle species that are often caught as bycatch in the SWIO. This will be done for both industrial fisheries (Chapter 3) and artisanal fisheries (Chapter 4).

Chapter 3

Bycatch of sea turtles in the longline and purse seine fisheries in the SWIO

Abstract

Bycatch of sea turtles in economically important industrial fisheries has received particular attention in recent years. However, very few studies have evaluated the scale of impact of industrial fisheries on bycaught populations/species. It is imperative to understand the effects of these industrial fishing activities before designing appropriate management options to achieve ecosystems-based fisheries management and thus reducing unnecessary risk to threatened species. This study aimed to quantify the interactions and mortality of the sea turtle species in industrial longline and purse seine fisheries within the SWIO region during 2000 - 2011. Publications, online databases and technical reports were used as sources of data to establish a database regarding the fishing effort and sea turtle bycatch in the region. The information contained in the subsequent database was used map the extent of fisheries and effort within the region and to quantify sea turtle bycatch per species in the respective fisheries. An estimated 4388 indiv.y^{-1} were caught, with an estimated direct mortality of 189 indiv.y^{-1}. The longline fishery was responsible for the bulk of these interactions with (mean ± SD) 4129 ± 1376 indiv.y^{-1} caught, and a direct mortality of 167 ± 53 indiv.y^{-1}. The most commonly caught species were loggerhead and leatherback turtles; with the greatest impact to the population recorded for leatherbacks, because the number of individuals caught relative to the nesting female population size was greater (1975.43 %) than that for loggerheads (173.26 %). The purse seine fishery caught 259 ± 34 indiv.y^{-1} (mean ± SD) with an associated direct mortality of 20 ± 2 indiv.y^{-1}. However, the impacts of FADs and post-release mortality were excluded from this estimate because of data limitations. Mitigation measures should be considered for the longline industry particularly concerning leatherback turtles in the SWIO because this small population cannot sustain the high bycatch rate.

Introduction

The Western Indian Ocean (WIO) region is globally one of the most important fishing grounds for skipjack tuna (*Katsuwonus pelamis*), and a principle fishing ground for yellowfin

(*Thunnus albacares*) and bigeye tuna (*Thunnus obesus*) (FAO, 2005). Here, up to 970 000 tonnes of tuna, with a processed value of € 2 - 3 billion, are harvested annually predominantly using purse seine and longline fishing methods (Ansell, 2006). The South Western Indian Ocean (SWIO) region also includes some of the world’s poorest countries (FAO, 2006), so these tuna fisheries are of significant social and economic importance to mainland and island states. Tuna and swordfish species (*Thunnus* spp., *Katsuwonus* spp., *Auxis* spp., *Scomberomorus* spp., *Ziphiass* spp., *Makaira* spp., *Tetrapturus* spp., and *Istiophorus* spp.) constitute some of the largest and most valuable marine food resources to these countries, and so contribute significantly to their GDPs. Foreign exchange is also generated through the receipts from fishing agreements, access/license fees, export of tuna and tuna products, as well as the expenditure by foreign fleets in the ports of the region (Barnes & Ansell, 2006). However, despite their significant economic value, there are concerns about the sustainability of these fisheries particularly with regards to the impacts on endangered mega fauna such as sea turtles caught as bycatch (Hall et al., 2000, Lewison & Crowder, 2007).

The life history of sea turtles (see Chapter 2) makes them particularly vulnerable to capture in fisheries for two reasons. First, sea turtles are long lived, and second they mature late, and hence have slow population growth rates (Heppell *et al.*, 2005). Population growth depends on the survival of a substantial fraction of individuals to reproductive maturity (Heppell, 1998). Thus even low levels of bycatch can have a significant impact on population growth rate if bycatch removes a substantial portion of the sea turtle population (Wallace *et al.*, 2010a), or if older (reproductively more valuable) age classes are more frequently caught (Crouse *et al.*, 1987, Heppell, 1998). The latter is of particular concern as larger sea turtles, which are also generally older, are more susceptible to capture in fishing gear. The impacts are also expected to be specific to the fishing activity, as these tend to select for specific size classes. Longlines frequently catch large individuals (Caminas *et al.*, 2006, Kotas *et al.*, 2004, Pinedo & Polacheck, 2004) whereas purse seine fishing operations may be less destructive as they tend to catch juveniles and sub-adults rather than adults sea turtles, and have lower mortality rates (Clermont *et al.*, 2012).

Industrial fisheries impacts on bycaught species frequently appear to be low because interactions between individual vessels and species such as sea turtles are rare. Further, most of these interactions are (initially) non-fatal and sea turtles or mammals are released alive. However, the extent of fisheries, which span entire ocean basins with multiple vessels operating in the same space over time, result in substantial cumulative effects (Wallace *et al.*, 2010a). Industrial fisheries are thus a major source of mortality in non-target species such as sea turtles (Casale, 2011, Lewison *et al.*, 2004b). Given these high rates of accidental capture, industrial fisheries can drive populations close to extinction (Spotila *et al.*, 2000).

The level of interaction between sea turtles and large-scale industrial fisheries and the possible impact that these have on populations in the SWIO have not yet been quantified. This knowledge gap is important to fill before possible management actions can be implemented to enhance the sustainability of these fisheries. Most studies have focussed on limited geographic distributions (e.g., single countries) or on single fleets (Petersen *et al.*, 2009, Poisson & Taquet, 2000). However, this does not reflect impacts at a population or regional management unit (RMUs) scale.

Given that all species of sea turtles are listed as threatened by the IUCN (IUCN, 2014) but the population sizes and distribution of sea turtle species differ, it is imperative to quantify the level of threat and the extent of overlap between industrial fisheries and sea turtles. Therefore, the aim of this chapter is to quantify the bycatch of sea turtles in the industrial longline and purse seine fishing activities across the entire SWIO, and to determine if bycatch rates are related to the extent of overlap between fisheries and sea turtles. To achieve this aim, there are three key objectives. First, to quantify the extent of fishing operations (in terms of effort and spatial extent) for both longline and purse seine fisheries. Second, to quantify the total number of each sea turtle species caught as bycatch in both longline and purse seine fisheries and the concomitant mortality rates for each species in both of the fisheries. Third, to determine the vertical and horizontal overlap between fishing operations and the known distribution of sea turtles as a possible explanation for the patterns observed.

Methods

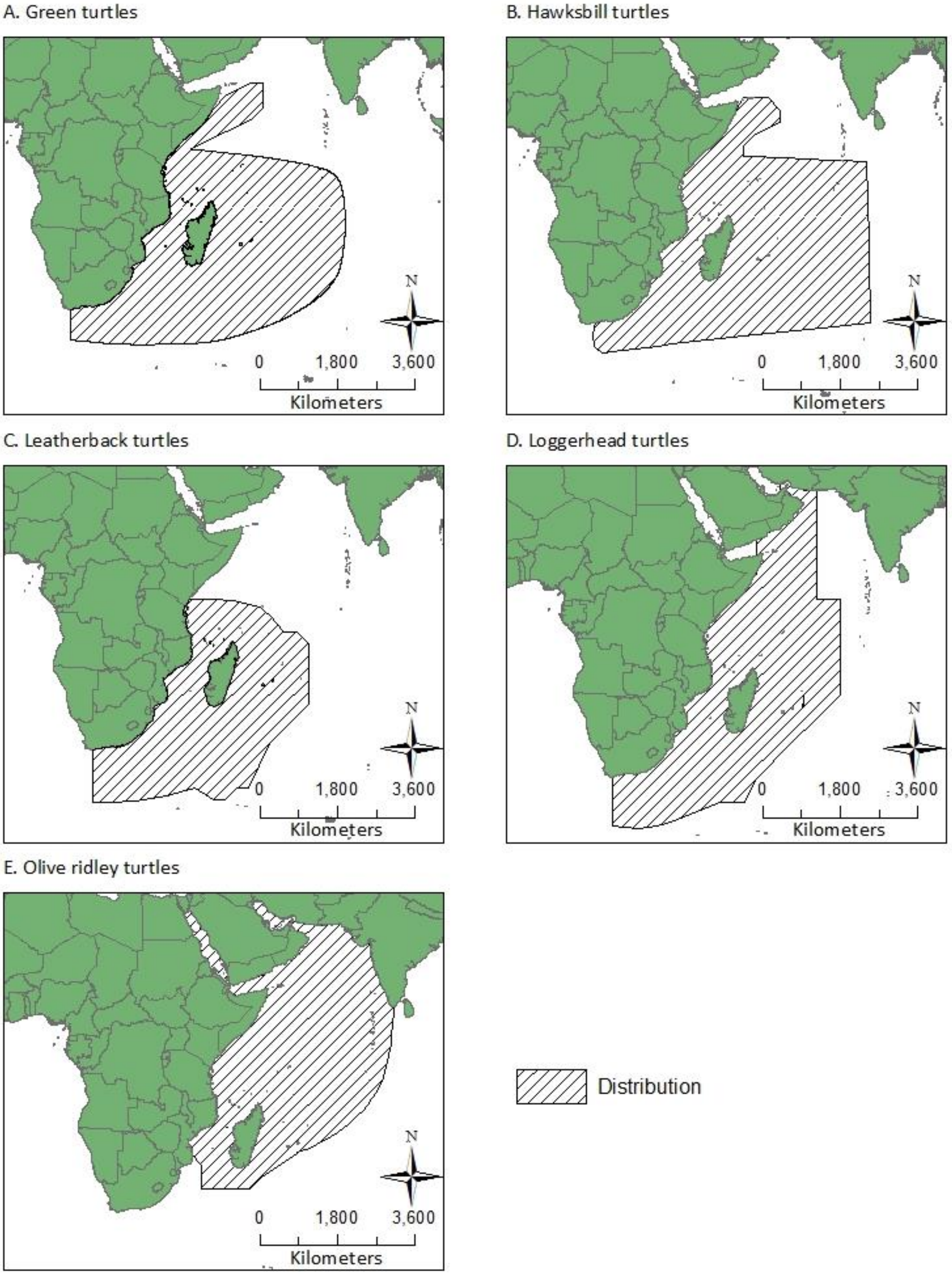
Fishery distribution

Data on longline and purse seine fishing distribution, and effort in the Indian Ocean were gathered from the Indian Ocean Tuna Commission (IOTC) online database (IOTC, 2013a, IOTC, 2013b). Catch data for longlining were available for the following countries: Australia, China, France, India, Japan, Mauritius, Portugal, Republic of Korea, Seychelles, Spain, South Africa, Taiwan, and United Kingdom. For the period from 2000 - 2011, several different metrics were used to report effort in the IOTC database. Of the 36 983 data entries for the specific period, 92.4 % of the effort data were reported using number of hooks set, 4.9 % as number of days at sea, 2.3 % as number of fishing days, and 0.4 % as number of sets performed. It was not possible to convert between metrics used for reporting so the analysis used data in the most commonly used metric i.e. fishing effort by the number of hooks set. The resolution of the data also varied between 1˚x1˚ grid squares and 5˚ x 5˚ grid squares. Therefore, data were plotted in ArcMap 10.2 (ESRI, 2013) as the effort per 2.5° x 2.5° grid square by aggregating finer scale data (1° x 1°) and parsing larger cells (5° x 5°) evenly. The at-sea distribution of sea turtle species that nest in the SWIO region span across the Western Indian Ocean (WIO) region (see Chapter 2), for this reason fisheries data for the entire WIO region were used.

Catch and effort data for purse seine fishing during the period 2000 - 2011 were available for France, Mauritius, Seychelles, Spain, as well as other unidentified fleets. Several different metrics were also used to report fishing effort (in the IOTC database) on purse seine fishing activities. Of the 135 703 data entries, 89.3 % reported fishing effort as number of fishing hours, 8.0 % as number of fishing days, 1.3 % as number of days at sea, 0.7 % as number of hours at sea, 0.4 % as number of trips, and 0.3 % as number of sets performed. As for longlining, only data in the most commonly reported effort metric were used, i.e., data as number of fishing hours. The resolution of all data reported was in 1° x 1° grid squares, so no modifications to the resolution were required and data were plotted in ArcMap 10.2 (ESRI, 2013).

Bycatch calculations

The sea turtle species distributions maps were created by combining the RMU distribution maps (Fig. 1.2) as per Wallace *et al.* (2010a) with all satellite tracking data available per RMU (see Chapter 2). The resultant maps (Fig 3.1) were overlaid on fishery effort maps in ArcMap 10.2 (ESRI, 2013), and the effort data in the region of overlap were extracted. Using these data, bycatch estimates for each of the fisheries were calculated as described below.


A. Green turtles

B. Hawksbill turtles

C. Leatherback turtles

D. Loggerhead turtles

E. Olive ridley turtles

Figure 3.1 Distribution maps for sea turtle species that nest within the SWIO; Distributions created from published and unpublished satellite tracking data (Appendix B) and RMU ranges as per Wallace *et al.* (2010a).

Longline fishery

The bycatch rates (most commonly reported as number of individuals caught per 1000 hooks) and the percentage mortality of sea turtles caught, were obtained from published literature and the IOTC country reports. Total annual bycatch for the longline industry (B_{TL}), as indiv.y^{-1} was estimated using the mean observed bycatch rate as indiv.1000 hooks^{-1} (B_{RL}) and the annual fishing effort (hooks) (E_L) per year *i*, for the total number of years (N_Y) from 2000 – 2011 (n = 12) as follows (Witzell, 1984):

$$\boldsymbol{B_{TL} = \frac{\sum_{i=1}^{n} B_{RL} \times E_{Li}}{N_Y}} \quad \textbf{Equation 3.1}$$

The total mortality per year (M_{TL}) was calculated as follows:

$$\boldsymbol{M_{TL} = B_{TL} \times M_{\%}} \quad \textbf{Equation 3.2}$$

where $M_{\%}$ is, the average percentage observed mortality during capture.

Purse seine fishery

For the purse seine fishery only one set of bycatch estimates was available. These were for the EU fleets from the French and Spanish observer programs (Bourjea *et al.*, 2014). The bycatch estimates of the report distinguished between sets on FADs (Fish Aggregation Devices) and FSC (free-swimming schools). However, effort data from the IOTC database does not clearly differentiate between the two methods. Thus, for the purposes of this study, no discrimination was made between bycatch of sets on FADs and FSCs, and so only one bycatch rate was calculated for the fishery. Bycatch rate in Bourjea *et al.* (2014) was reported as bycatch per set, but the majority of effort data from the IOTC database were reported as fishing hours. A conversion (between number of sets and fishing hours) was thus necessary. This was calculated by extracting the total effort of French and Spanish fleets from the IOTC database for the period between 2003 – 2010 corresponding to the study of Bourjea *et al.* (2014). Using the extracted data from the IOTC database and the estimated mean number of sea turtles caught per year (N_T) (in this case 250 sea turtles per year), a mean bycatch rate (B_{RP}) as indiv.fishinghour^{-1} was calculated using the following equation:

$$\boldsymbol{B_{RP} = \frac{N_T}{F_H}} \quad \textbf{Equation 3.3}$$

where F_H is the mean number of fishing hours per year for the French and Spanish fleets in the IOTC database for the period between 2003 – 2010.

This estimate provided a bycatch rate per number of fishing hours. Total bycatch per year for the purse seine fishery (B_{TP}) as indiv.y^{-1} and number of mortalities per year (M_{TP}, indiv.y^{-1}) was estimated using the annual fishing effort (fishing hours, E_P) per year i for the total number of years (N_Y) from 2000 – 2011 (n = 12), for all fleets reported in the IOTC database and the calculated bycatch rate using the following equations:

$$\boldsymbol{B_{TP} = \frac{\sum_{i=1}^{n} B_{RP} \times E_{Pi}}{N_Y}}$$ **Equation 3.4**

$$\boldsymbol{M_{TP} = B_{TP} \times M_{\%}}$$ **Equation 3.5**

where $M_{\%}$ is, the average percentage observed mortality during capture.

Spatial overlap between fisheries and sea turtles

The area (km^2) of each of the sea turtle distributions per species (Fig. 3.1; D_T) was determined in ArcMap 10.2 (ESRI, 2013). Thereafter the extent of each fishery generated above (independent of intensity) that fell within the distribution of the sea turtle species (D_F) was determined. The percentage horizontal overlap (O_H) between each of the species and the fishery was then calculated using the following equation:

$$\boldsymbol{O_H = \frac{D_F}{D_T} \times 100}$$ **Equation 3.6**

with D_F being the spatial extent (km^2) of sea turtle distribution where fishing activity occurs, and (D_T) being the total spatial distribution (km^2) of sea turtles.

In order to determine the vertical overlap between fisheries and sea turtle species (O_V), firstly the maximum diving depth (in m) of sea turtle species (DD_T) were determined (Tables 2.1 to 2.5). Secondly the operational fishery depth (OD_F, Fig. 3.2) that occurs within the diving depth range of sea turtle species were determined. This was done by bounding the operational fishing depth to the diving depth range (thus, where the depth range of a fishery exceeds the depth range of sea turtle species ($DF_{max} > DD_T$) the depth range was limited to the depth range of sea turtle species ($DF_{max} = DD_T$) using the following equation:

$$\boldsymbol{OD_F = \min(DF_{max}, DD_T) - DF_{min}}$$ **Equation 3.7**

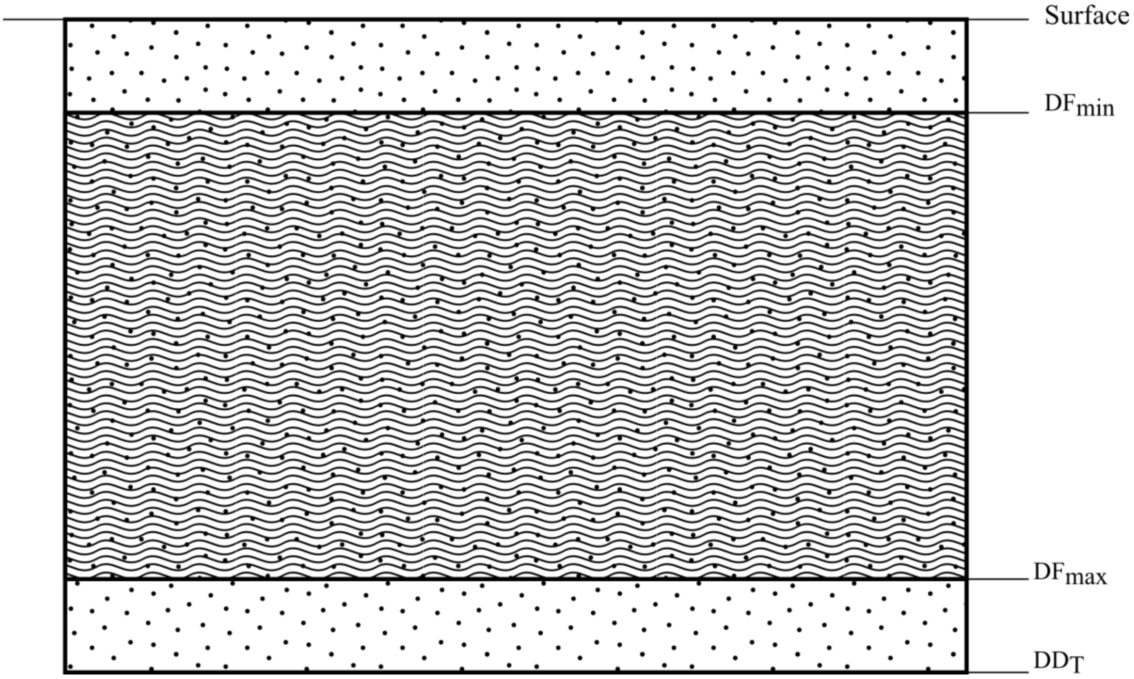


Figure 3.2 Illustration of calculation of the vertical overlap of fisheries with the diving depth of turtles showing turtle diving distribution in dots and operational depth range of the fishery in waves.

Where DF_{max} is the maximum depth range of the fishery and DF_{min} is the minimum depth range of the fishery.

$$\boldsymbol{O_V = \frac{OD_F}{DD_T} \times 100} \qquad \textbf{Equation 3.8}$$

Vertical overlap calculations for the longline fisheries were done for the operational range of both shallow-set (15 - 100 m) and deep-set (30 - 400 m) gear (Beverly *et al.*, 2003). For purse seine fishing, vertical overlap was calculated for the operational range of both the maximum depth range (0 - 300 m) targeted by purse seine fisheries (Hall & Roman, 2013) as well as the most commonly targeted depth range (0 - 70 m, IOTC, 2010). Vertical overlap was calculated for both the maximum diving depth of sea turtles and the mean diving depth of sea turtles. Mean diving depth (D_M, in meters) was calculated for each species as the weighted average of the mean depths (d) reported in published literature (weighted by the number of sea turtles used per study) using the following equation:

$$\boldsymbol{D_M = \frac{\sum_{i=1}^{n} c_i d_i}{\sum_{i=1}^{n} c_i}} \qquad \textbf{Equation 3.9}$$

Where c_i is the number of sea turtles used in the ith study, and n is the number of studies used per species (green turtles n =16, hawksbills n = 9, leatherback n = 15, loggerheads n = 5, olive ridleys n = 1).

With weighted standard deviation (sd_w^2) calculated using the following equation:

$$\boldsymbol{sd_w^2} = \sqrt{\frac{\boldsymbol{N'\sum_{i=1}^{N} c_i(d_i - D_M)^2}}{\boldsymbol{(N'-1)\sum_{i=1}^{N} c_i}}} \qquad \textbf{Equation 3.10}$$

where N is the number of studies and N' is the number of studies with non-zero weights.

To determine whether bycatch per species was related to the degree of overlap the bycatch data were tested for normality using the Shapiro Wilk test (Shapiro & Wilk, 1965). All data fit the assumption of normality, and Pearson product moment correlations (Pearson, 1901) were performed to assess the correlation between overlap and the magnitude of bycatch. All calculations were done in Excel and statistical analysis were performed in R (R Core Team, 2014).

Results

Fishery distribution, effort and bycatch

Longline fisheries

Nations fishing for tuna and swordfish in the WIO include China, India, Japan, Korea, Mauritius, Portugal, Reunion, Seychelles, South Africa, Spain, Taiwan, and United Kingdom. Longline fishing effort (2000 – 2011) was distributed throughout 36 813 645 km^2 of the WIO, but with great variation in the distribution of effort in both space and time (inter-annual differences) across the region (Fig. 3.3). Hotspots were identified by combining data over time. These hotspots were off the Horn of Africa and south of the Mozambique Channel (Fig. 3.4). The quantitative impacts are reported as mean ± SD (standard deviation) unless stated otherwise. The total number of fishing hooks set in the WIO region for the period from 2000 to 2011 was 3 085 924 708 hooks, averaging 257 160 392.4 ± 85 274 464.6 hooks.y^{-1}. The number of hooks per 2.5° x 2.5° square ranged from 1 100 - 36 256 997 hooks, with a mean of 6 038 992 ± 6 710 429 hooks.square^{-1}. Total reported fishing effort peaked in 2005 in the WIO and gradually declined thereafter (Fig. 3.5).

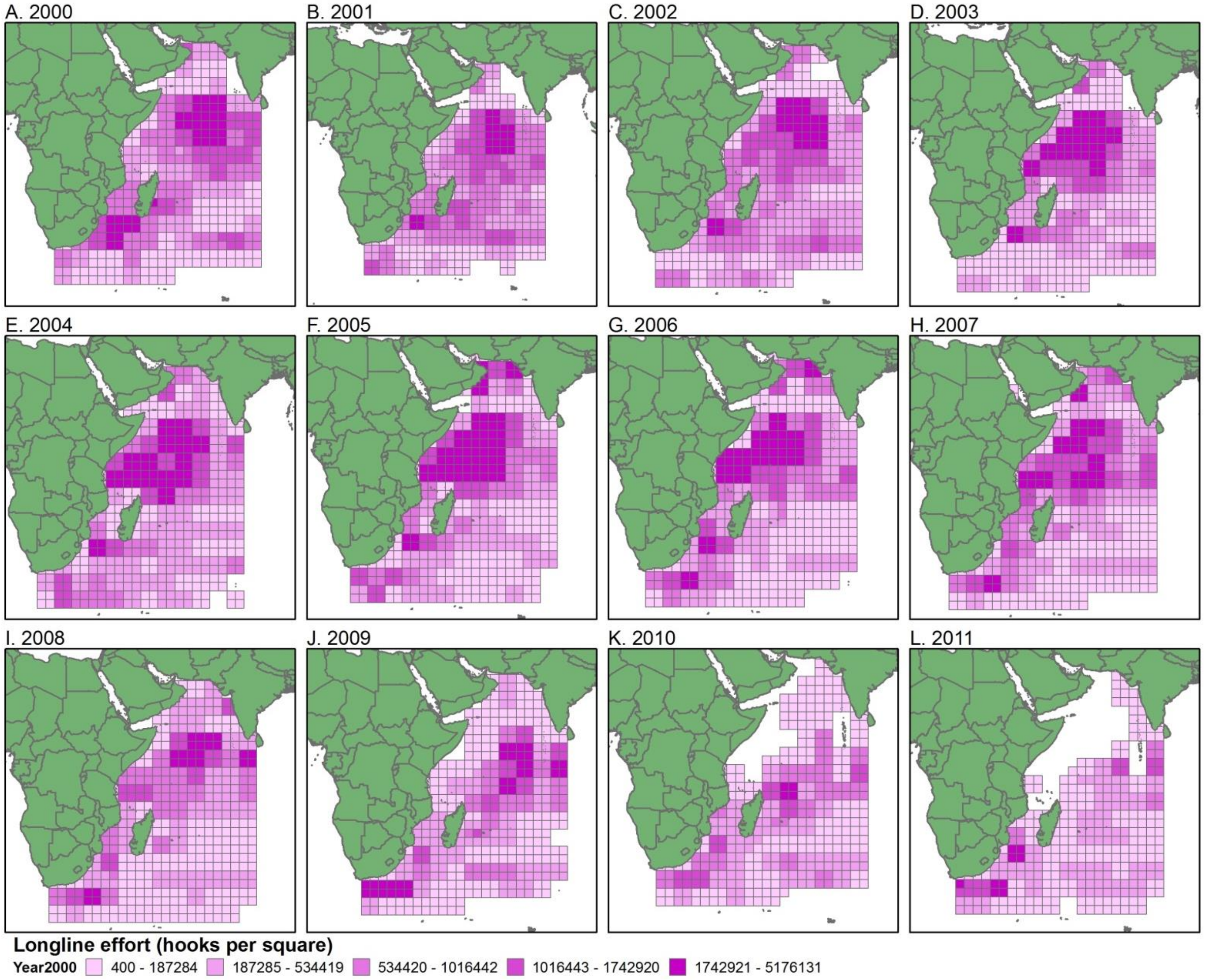


Figure 3.3 Annual distribution and effort (hooks set per 2.5° x 2.5° square) of longline fisheries in the WIO from 2000 to 2011.

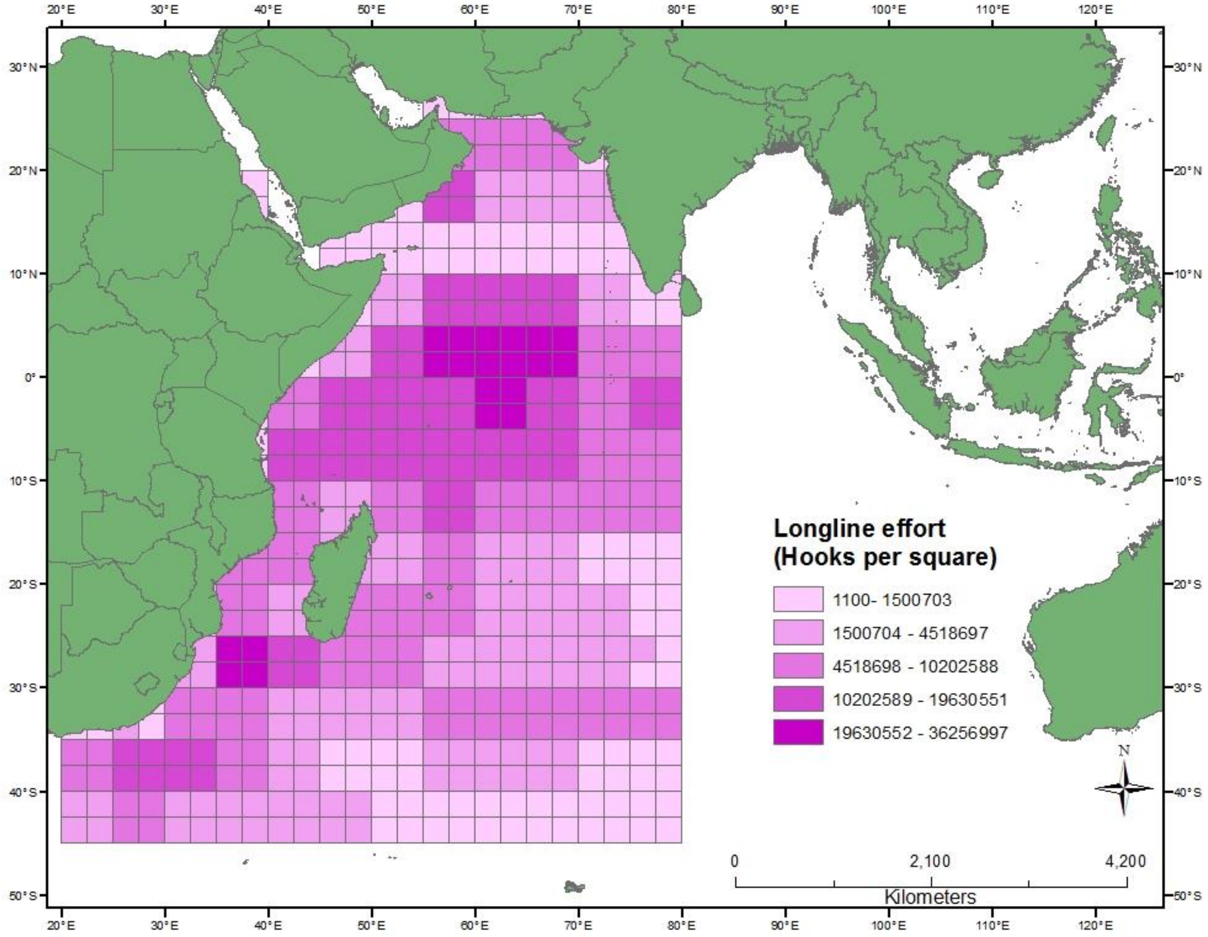


Figure 3.4 Distribution and total longline fishing effort (in number of hooks set) per 2.5° x 2.5° square in the WIO for the period 2000 – 2011.

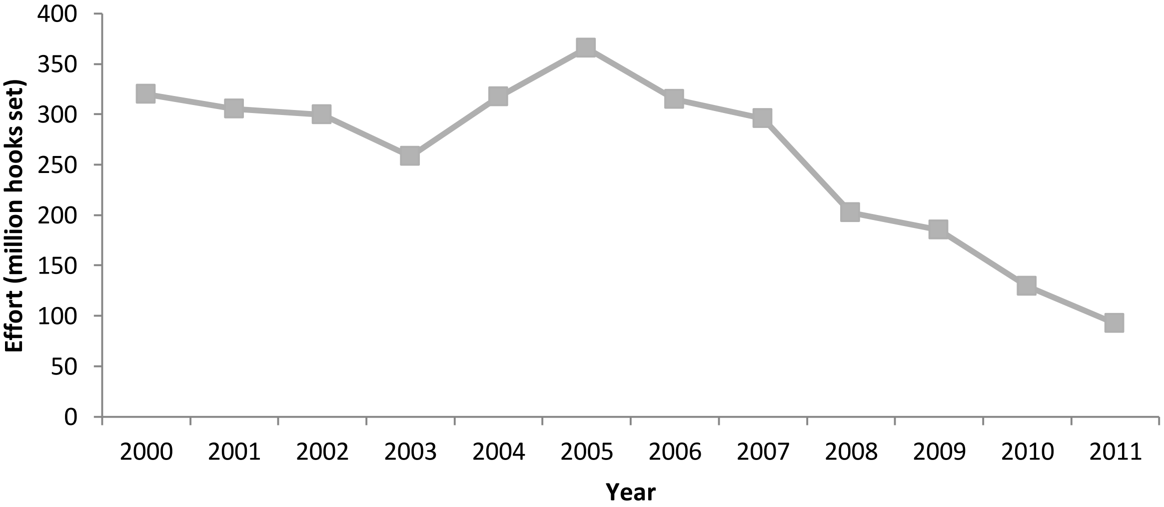


Figure 3.5 Total reported fishing effort for the longline industry per year (as millions of hooks set) for the period from 2000 to 2011 in the WIO, showing a decline in the reported effort since 2005.

Even though target catch and effort information was readily available, few data were available for bycatch of sea turtles. The bycatch rates of the available data varied considerably (Table 3.1). The highest mean capture rate was observed for leatherbacks (0.0121 ± 0.0113 indiv.1000 hooks^{-1}) and loggerheads (0.0082 ± 0.0070 inidv.1000 hooks^{-1}). The capture rate of leatherback turtles was thus more than one and a half times larger than that for loggerheads and eight times larger than that for hawksbill and green turtles, which had the lowest mean capture rate (0.0015 ± 0.0015 indiv.1000 hooks^{-1}). The mortality of captured individuals varied among species, and ranged from 0 % for olive ridley turtles to 13.6 ± 18.2 % for green turtles. Green turtles seem reasonably vulnerable to mortality, with the percentage of captured sea turtles dying being almost three and a half times higher than that of loggerhead and leatherback turtles.

The longline fishery in the SWIO captured an average of 4128.93 ± 1375.56 indiv.y^{-1} (Table 3.2 and Fig. 3.6) for the time period investigated. The average number of individuals caught per species ranged from 311.43 ± 101.68 indiv.y^{-1} for green turtles to 1697.90 ± 620.30 indiv.y^{-1} for loggerheads. Leatherbacks are the second most commonly caught species with 1422.31 ± 371.56 leatherbacks caught per year. In order to compare the bycatch to the population size, the number of individuals per species caught was expressed as a percentage of the number of nesting females, because the only comparative estimates available for population sizes are those of nesting females. These percentages vary among species. It was lowest for green turtles at 2.94 % and highest for leatherbacks at 1975.43 % (Table 3.2). A Shapiro-Wilk test indicated that normality is a reasonable assumption for the longline bycatch data ($W = 0.862$, $p = 0.2353$), thus a Pearson product-moment correlation coefficient was computed to assess the relationship between the annual bycatch rates and the abundance of species. There was no significant correlation between the two variables, ($r = 0.257$, $p = 0.632$ $n = 5$), i.e. the bycatch rate and the abundance of the species in the region.

The longline fishery is responsible for the mortality of 167.21 ± 53.08 indiv.y^{-1} (Table 3.2 and Fig. 3.6) for the time period investigated. The number of mortalities per species per year ranged from 0 for olive ridleys to 67.92 ± 24.81 indiv.y^{-1} for loggerheads. The number of mortalities expressed as percentage of the number of annual nesting females ranged from the lowest (0 %) for olive ridleys to 79.01 % for leatherback turtles (Table 3.2).

Table 3.1 Summary of available bycatch rates ($\boldsymbol{B_{RL}}$) and percentage mortality ($\boldsymbol{M}_{\%}$) for the SWIO that were used in the calculation of the average bycatch rates per turtle species (indiv.1000hooks^{-1}), and average mortality (as percentage mortality of individuals caught). Where no data were available it is flagged as data deficient (DD).

Fleet	**France – EU (2007 - 2009)**		**France – Reunion (1997 - 1999)**		**South Africa (1998 – 2005)**		**South Africa (2006 – 2012)**		**Spain – EU (2005)**		**Mean rates ± SD**	
	(Chavance *et al.*, 2010)		(Poisson & Taquet, 2000)		(Petersen *et al.*, 2009)		(Unpublished data Oceans and Coasts)		(Ariz *et al.*, 2006)			
	$\boldsymbol{B_{RL}}$	$\boldsymbol{M}_{\%}$	$\boldsymbol{B_{RL}}$	$\boldsymbol{M}_{\%}$	$\boldsymbol{B_{RL}}$	$\boldsymbol{M}_{\%}$	$\boldsymbol{B_{RL}}$	$\boldsymbol{M}_{\%}$	$\boldsymbol{B_{RL}}$	$\boldsymbol{M}_{\%}$	**Bycatch Rate**	**Mortality rate**
Green turtles	0	0	0.0024	38.46	0.001	16	0.0005	DD	0.0038	0	0.0015 ± 0.0015	13.6 ± 18.2
Hawksbills	0	0	0.0068	37.84	0.001	16	0	0	0	0	0.0016 ± 0.0030	10.8 ± 16.6
Leatherbacks	0.006	0	0.0073	0	0.01	16	0.0052	DD	0.0320	0	0.0121 ± 0.0113	4.0 ± 8.0
Loggerheads	0.012	0	0	0	0.018	16	0.0074	DD	0.0038	0	0.0082 ± 0.0070	4.0 ± 8.0
Olive ridleys	0	0	0	0	0	0	0.0005	DD	0.0019	0	0.0039 ± 0.0084	0

Table 3.2 Mean annual bycatch (indiv.y^{-1}) and mean annual mortality (indiv.y^{-1}) per species in the longline fishery. Bycatch and mortality are also expressed as the percentage of the annual number of nesting females in the SWIO (WIO population size used for olive ridleys).

Species	**Females**	**Bycatch**		**Mortality**	
		Mean ± SD	**% of nesting females**	**Mean ± SD**	**% of nesting females**
Green turtles	10599	311.43 ± 101.68	2.94	42.40 ± 13.84	0.40
Hawksbills	2762	313.15 ± 91.79	11.34	33.73 ± 9.89	1.22
Leatherbacks	72	1422.31 ± 371.56	1975.43	56.89 ± 14.86	79.01
Loggerheads	980	1697.90 ± 620.30	173.26	67.92 ± 24.81	6.94
Olive ridleys	1114	697.20 ± 295.87	62.59	0	0.00
Total		**4128.93 ± 1375.56**		**167.21 ± 53.08**	

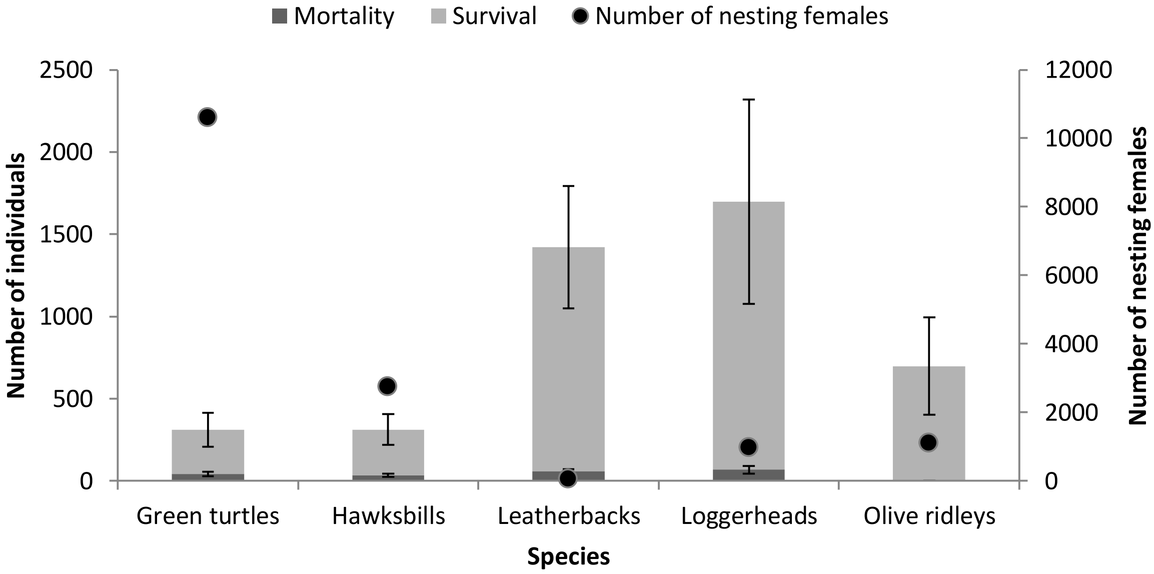


Figure 3.6 Number of turtles caught per year in the longline fishery in the WIO, indicating mortality, and survival per species as well as the annual number of nesting females per species per year. Error bars indicate standard deviation.

Purse seine fisheries

Countries with purse seine fishing fleets operating in the WIO region during 2000-2011, include France, Iran, Italy, Japan, Korea, Seychelles, Spain, Russia, and Thailand. Purse seine fishing in the Indian Ocean (during this period) was distributed throughout 12 748 583 km^2 of the WIO and was concentrated mainly around the island nations of Seychelles and Comoros in the northern and western sections of the region (Fig. 3.7). The purse seine fishing effort varied in space and time showing inter-annual differences (Fig. 3.8). All values are reported as mean ± SD (standard deviation) unless otherwise stated. The total number of reported fishing hours in the Indian Ocean by the purse seine fleets for the period 2000 to 2011 was 1 745 307 hours, averaging 145 442.2 ± 21 824.3 hours.y^{-1}. The number of fishing hours per 1° x 1° square ranged 2.5 – 14 680 hours with a mean number of 1 520.3 ± 2 289.6 hours per square. Reported fishing effort peaked in 2007 after which it gradually declined (Fig. 3.9).

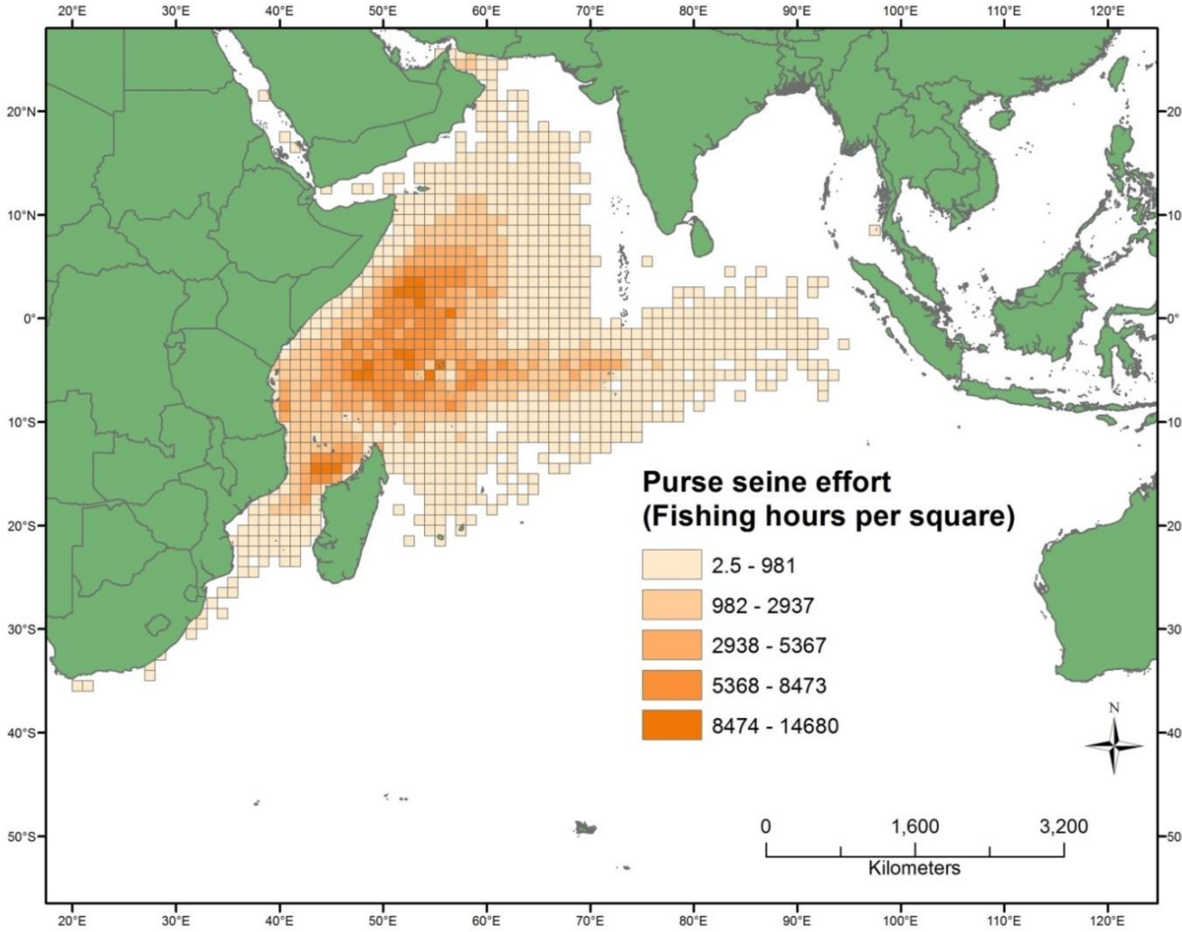


Figure 3.7 The distribution and total effort (in number of fishing hours) per 1° x 1° square for purse seine fisheries in the Indian Ocean for the period from 2000 – 2011.

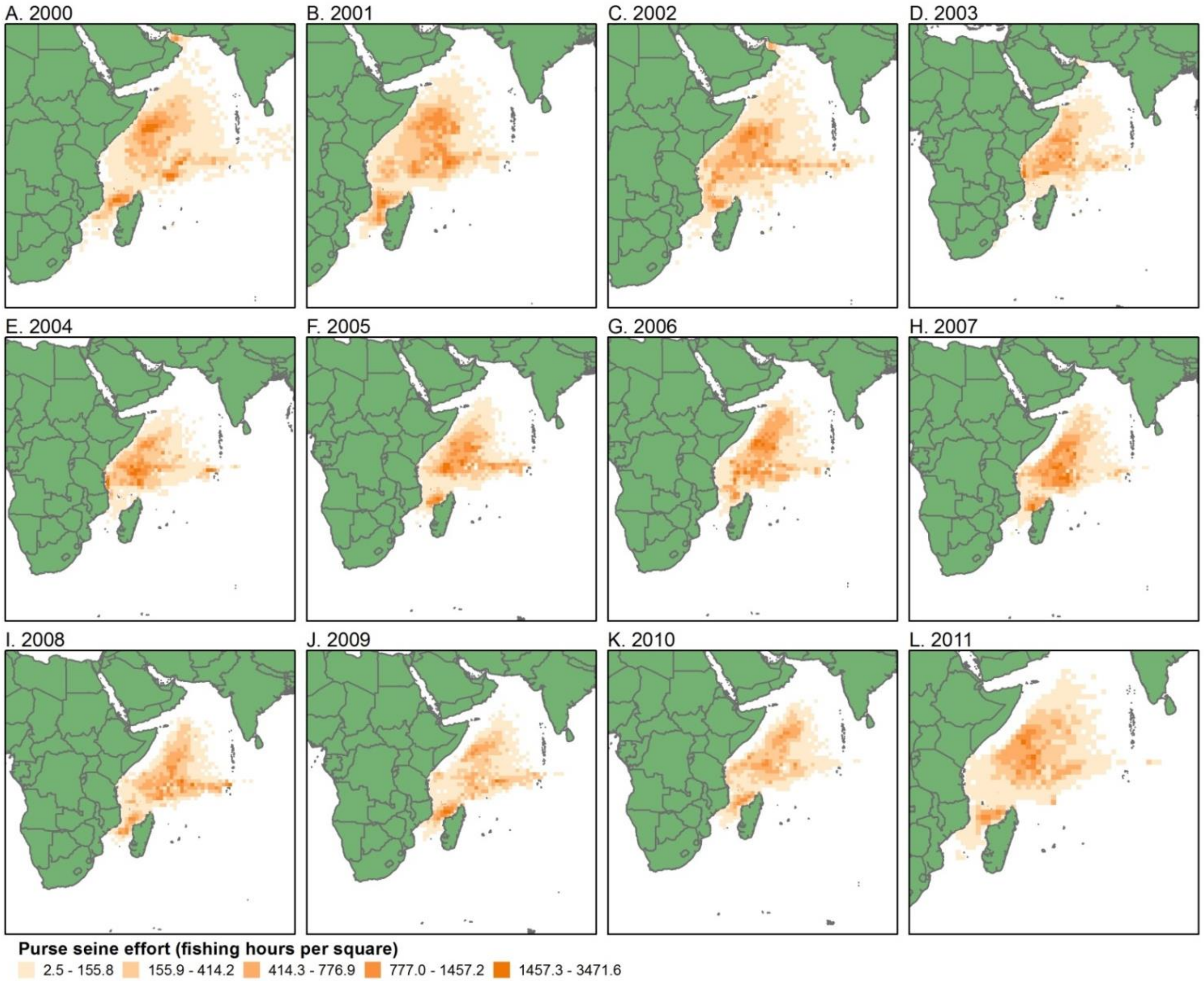


Figure 3.8 Annual distribution and effort (fishing hours per 1° x 1° square) of purse seine fisheries in the WIO from 2000 to 2011.

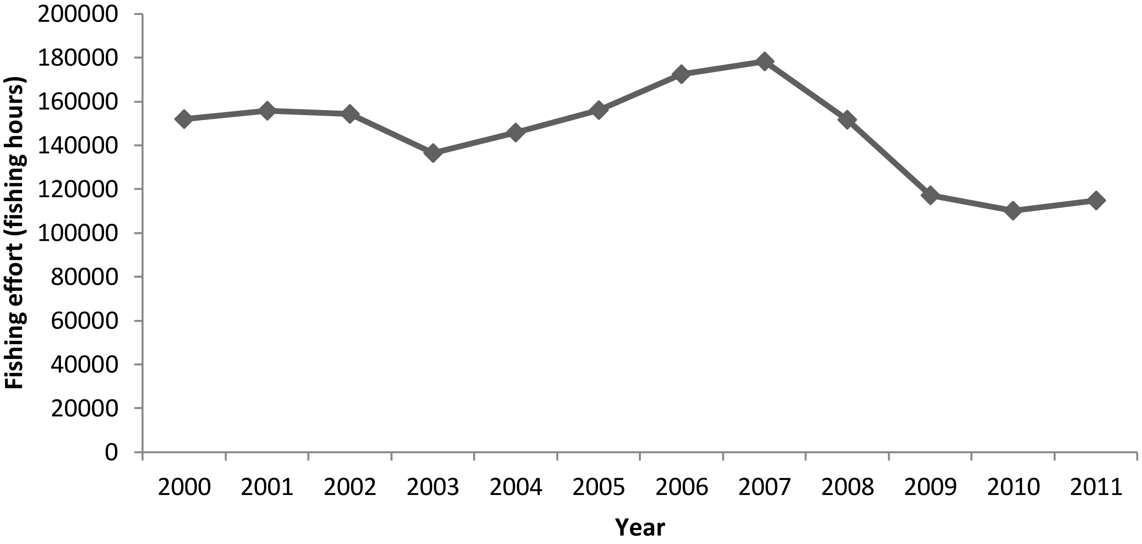


Figure 3.9 Total reported purse seine fishing effort per year (as fishing hours) in the WIO for the period from 2000 to 2011.

The mean bycatch rate calculated per 1000 fishing hours was 2.42 ± 0.46 indiv.1000 hours^{-1} with the bycatch rate per species ranging between 0.03 ± 0.01 indiv.1000 hours^{-1} for leatherbacks and 0.77 ± 0.15 indiv.1000 hours^{-1} for olive ridleys. The second highest capture rate was for hawksbills turtles at 0.49 ± 0.09 indiv.1000 hours^{-1} (Table 3.3 and Fig. 3.10). Multiplying these catch rates with the mean effort per year indicate a mean of 258.58 ± 33.72 indiv.y^{-1} caught in the purse seine industry in the SWIO. Mortality was calculated at 19.62 ± 2.13 indiv.y^{-1} (7.6 %; Table 3.4). The annual bycatch rate per species ranged between 1.74 ± 0.36 indiv.y^{-1} to 111.51 ± 17.01 indiv.y^{-1}, with the lowest bycatch rate for leatherback turtles (1.74 ± 0.36 indiv.y^{-1}) and the highest annual bycatch rate was recorded for olive ridleys turtles (111.51 ± 17.01 indiv.y^{-1}). The number of individuals caught per species as percentage of the annual number of nesting females ranged from 0.5 % of the green turtle population to 10.0 % of olive ridleys, with the second highest percentage for loggerheads (5.8 %; Table 3.4). A Pearson product-moment correlation coefficient was computed to assess the relationship between the bycatch rate and the abundance of the species. There was no significant correlation between the two variables ($r = 0.546$, $n = 5$, $p = 0.284$).

Table 3.3 Mean bycatch rates (indiv.1000hours^{-1}) and mortality (percentage of individuals caught) for the EU purse seine fleet in the Indian Ocean from 2003 – 2011.

Species	Bycatch rate (Mean ± SD)	Mortality (%)
Green turtles	0.43 ± 0.08	6.25
Hawksbills	0.49 ± 0.09	5.41
Leatherbacks	0.03 ± 0.01	0
Loggerheads	0.25 ± 0.05	15.79
Olive ridleys	0.77 ± 0.15	6.9

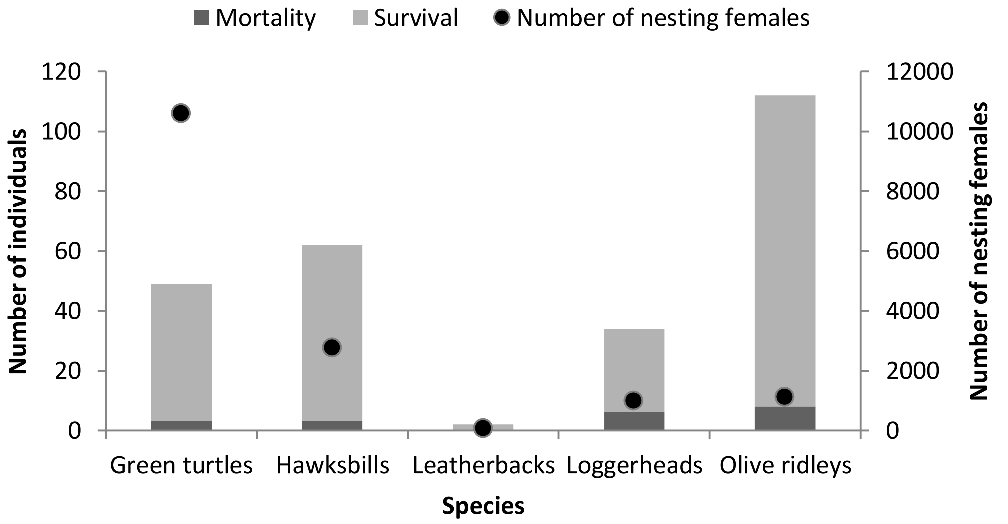


Figure 3.10 Number of sea turtles caught per year in the purse seine fishery in the WIO, indicating mortality, and survival per species as well as the annual number of nesting females per species per year.

Table 3.4 Mean annual bycatch rate (indiv.y^{-1}) and mean annual mortality rate (indiv.y^{-1}) per species for the purse seine fishery. Bycatch and mortality are also expressed as the percentage of the annual number of nesting females in the SWIO (WIO population size used for olive ridleys).

Species	Females	Bycatch		Mortality	
		Mean ± SD	% of nesting females	Mean ± SD	% of nesting females
Green turtles	10599	49.20 ± 7.81	0.46	3.08 ± 0.49	0.03
Hawksbills	2762	61.64 ± 9.90	2.23	3.33 ± 0.43	0.12
Leatherbacks	72	1.74 ± 0.36	2.42	0	0.00
Loggerheads	594	34.49 ± 5.33	5.81	5.52 ± 0.85	0.93
Olive ridleys	1114	111.51 ± 17.01	10.01	7.69 ± 1.17	0.69
Total		**258.58 ± 33.72**		**19.62 ± 2.13**	

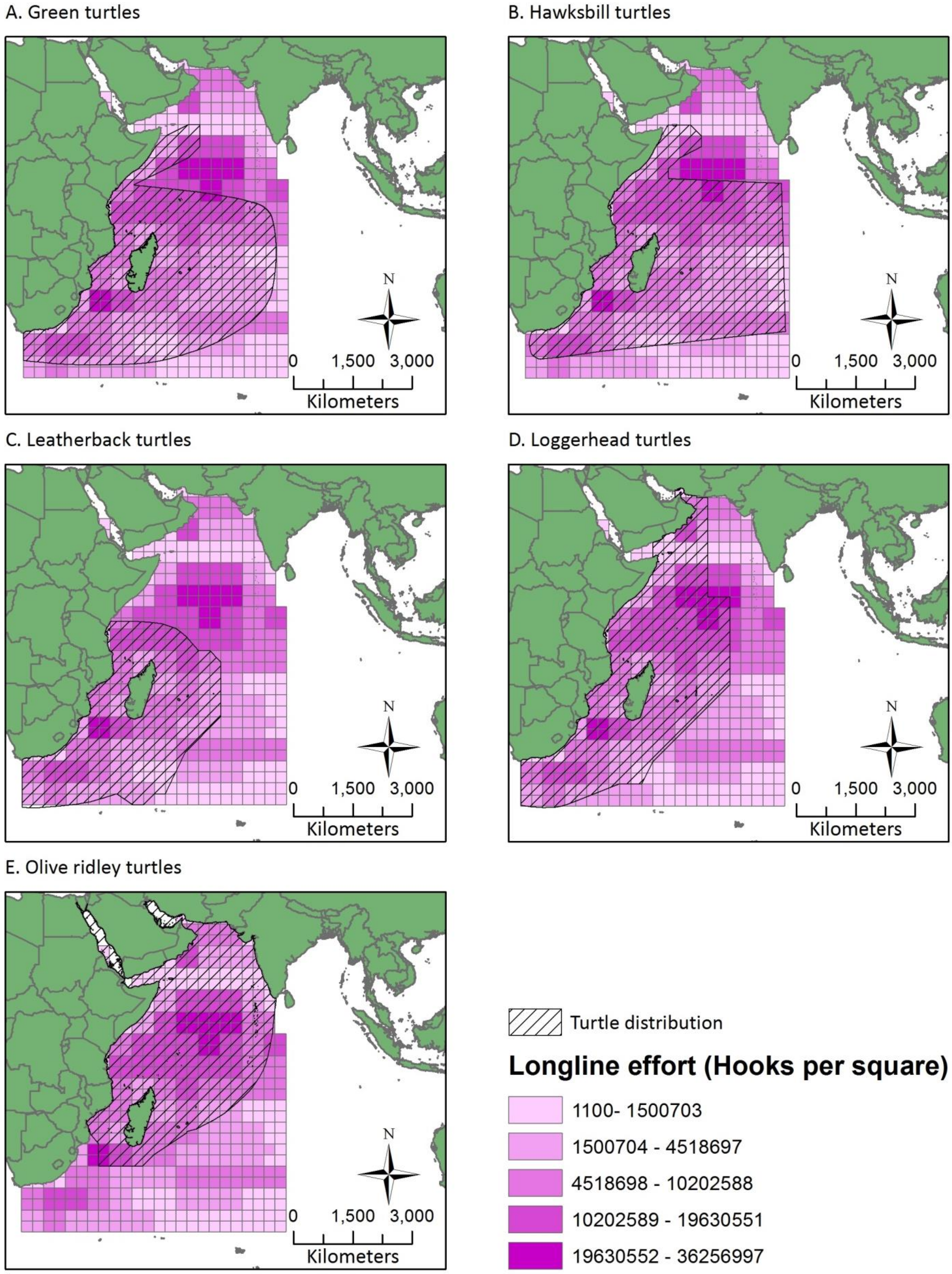
The total number of mortalities ranged between 0 indiv.y^{-1} for leatherbacks, to 7.69 ± 1.17 indiv.y^{-1} for olive ridleys. The mortality expressed as a fraction of the nesting female population size ranged between 0 % (leatherbacks) and 0.9 % (loggerheads), with olive ridleys as second highest (0.7 %). The large leatherbacks therefore seem reasonably immune to purse seine mortality whereas the smaller hard-shelled sea turtles (hawksbills, loggerheads, and olive ridleys) seem vulnerable.

Spatial overlap between industrial fisheries and sea turtle species

Horizontal overlap

There was substantial overlap (> 90 % in all cases) between the horizontal distribution of longline fisheries activities and distribution of all sea turtle populations in the SWIO (Fig. 3.11 and 3.12). A Shapiro-Wilk test indicated that normality is a reasonable assumption for the longline bycatch data ($W = 0.862$, $p = 0.2353$), and a Pearson product-moment correlation coefficient was computed to assess the relationship between the bycatch rate and horizontal overlap. There was no significant correlation between the two variables ($r = 0.842$, $n = 5$, $p = 0.052$; Fig. 3.13).

The horizontal overlap between purse seine fisheries and sea turtle distribution was far less than for the longline fishery. The overlap ranged among sea turtle species (see Fig. 3.12 and 3.14), with olive ridleys having the largest overlap (70 %) and leatherback turtles the smallest overlap (30 %). Purse seine bycatch data can reasonably assumed to be normal according to the Shapiro-Wilk test for normality ($W = 0.9764$, $p = 0.9147$). A Pearson product-moment correlation coefficient was computed to assess the relationship between the bycatch rate and horizontal overlap. There was a significant correlation between the two variables ($r = 0.92$, $n = 5$, $p = 0.018$). Overall, there was a strong, positive correlation between bycatch rate and horizontal overlap. Increase in percentage horizontal overlap was correlated with increase in annual bycatch rate (Fig. 3.15).


A. Green turtles

B. Hawksbill turtles

C. Leatherback turtles

D. Loggerhead turtles

E. Olive ridley turtles

Figure 3.11 Total longline distribution and effort (hooks set per 2.5º x 2.5º square) in the WIO between 2000 – 2011 showing horizontal overlap with sea turtle species.

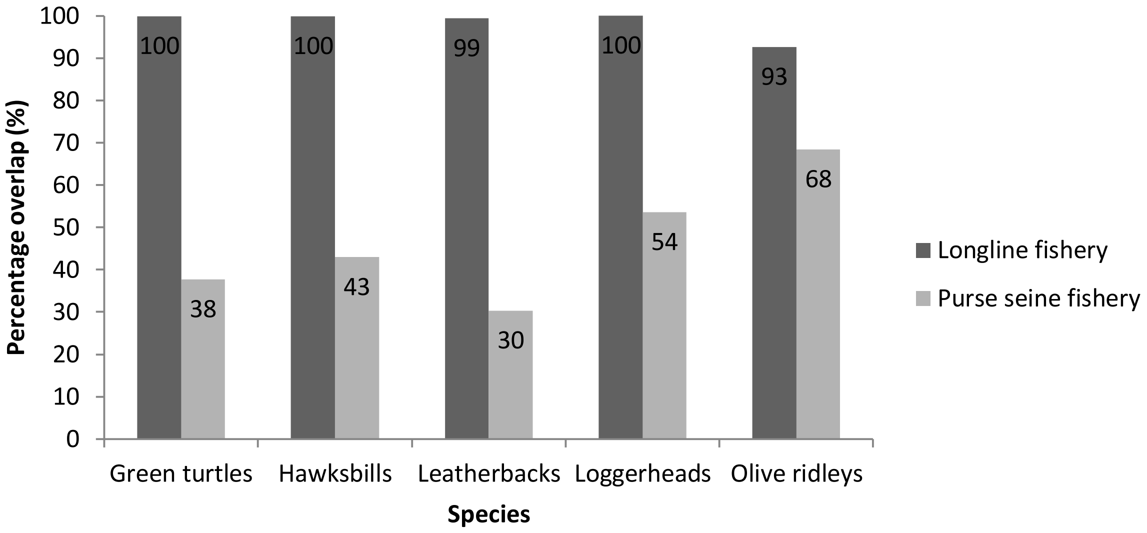


Figure 3.12 The percentage horizontal overlap (%) between the spatial distribution of longline and purse seine fisheries in the WIO, and distribution of sea turtle species.

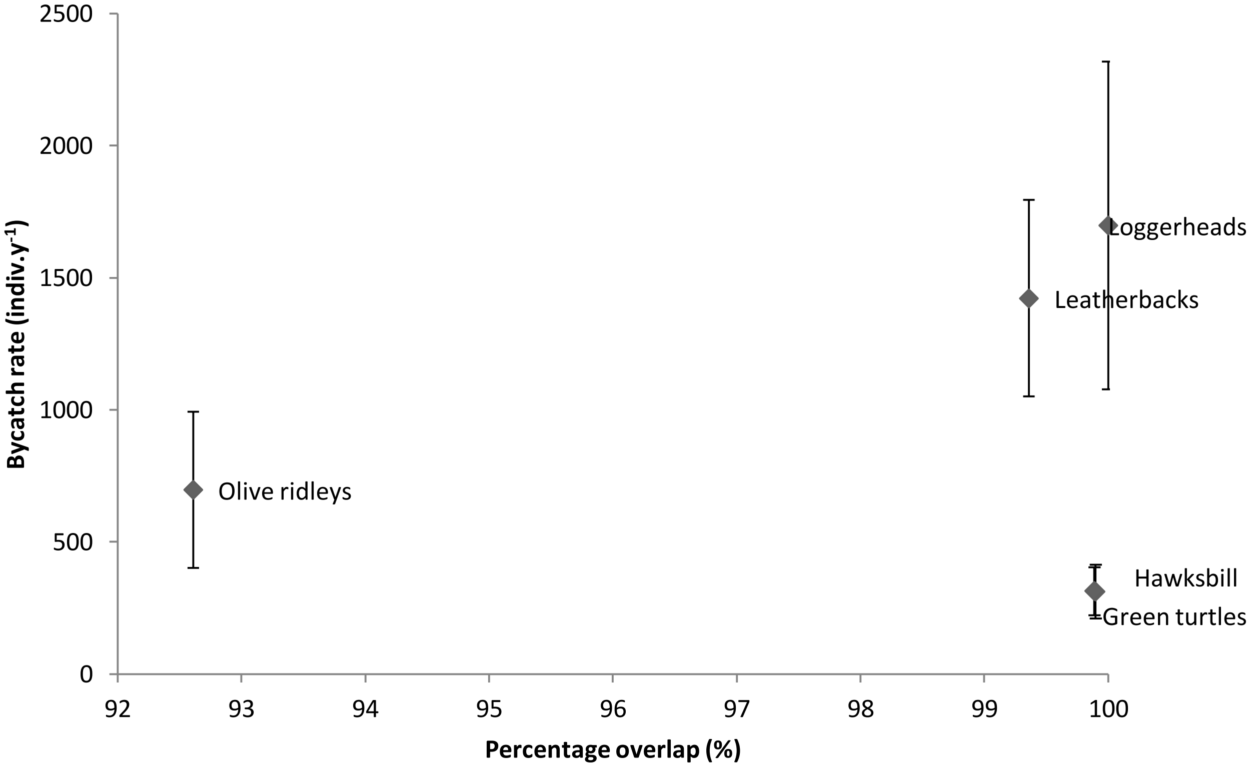
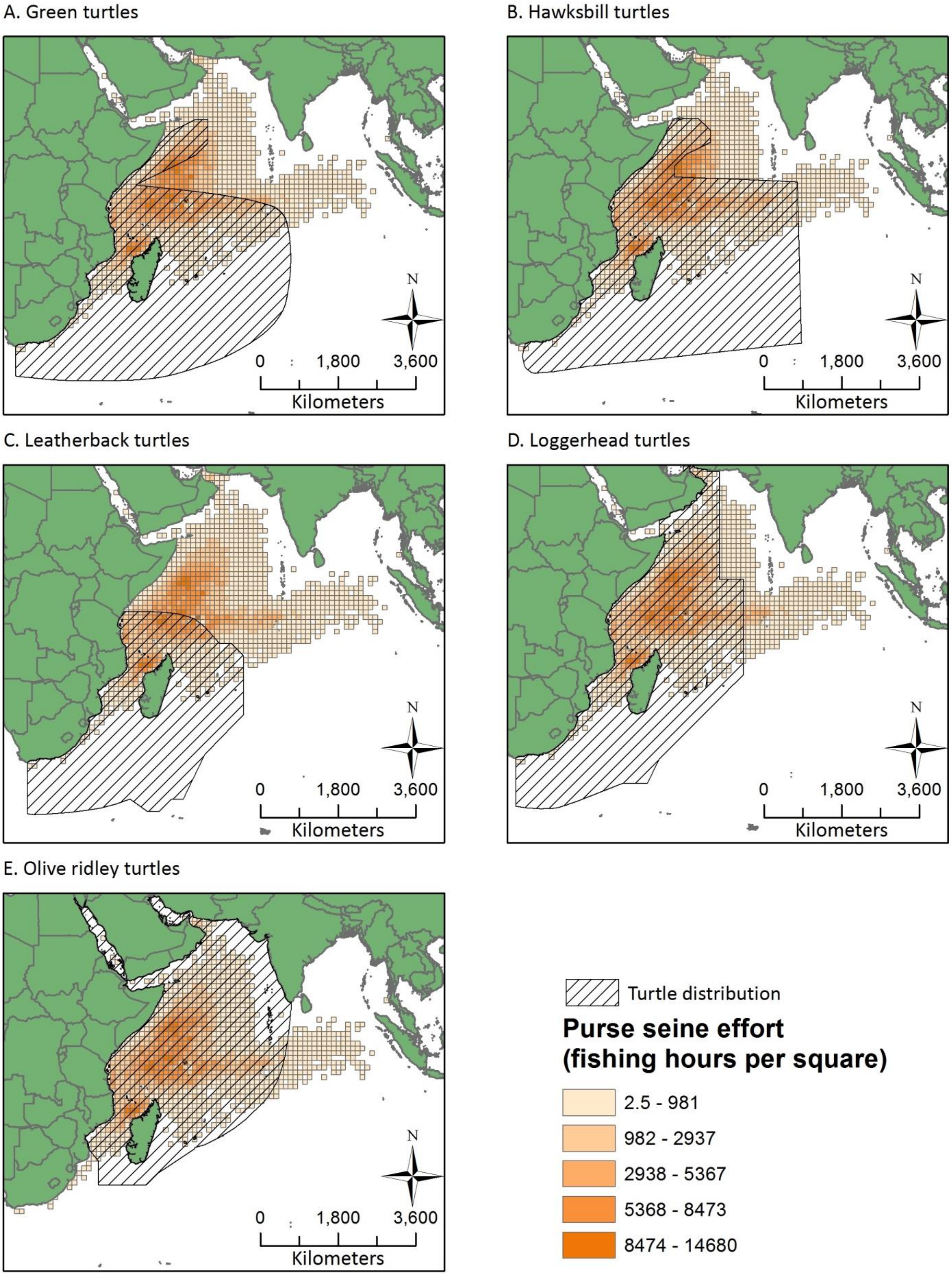


Figure 3.13 Bycatch rate (indiv.y^{-1}) per species expressed as a function of percentage horizontal overlap with the longline fishery in the WIO. Data are presented as the mean ± standard deviation.


A. Green turtles

B. Hawksbill turtles

C. Leatherback turtles

D. Loggerhead turtles

E. Olive ridley turtles

Figure 3.14 Total purse seine distribution and effort (fishing hours per 2.5º x 2.5º square) in the WIO between 2000 – 2011 showing horizontal overlap with sea turtle species.

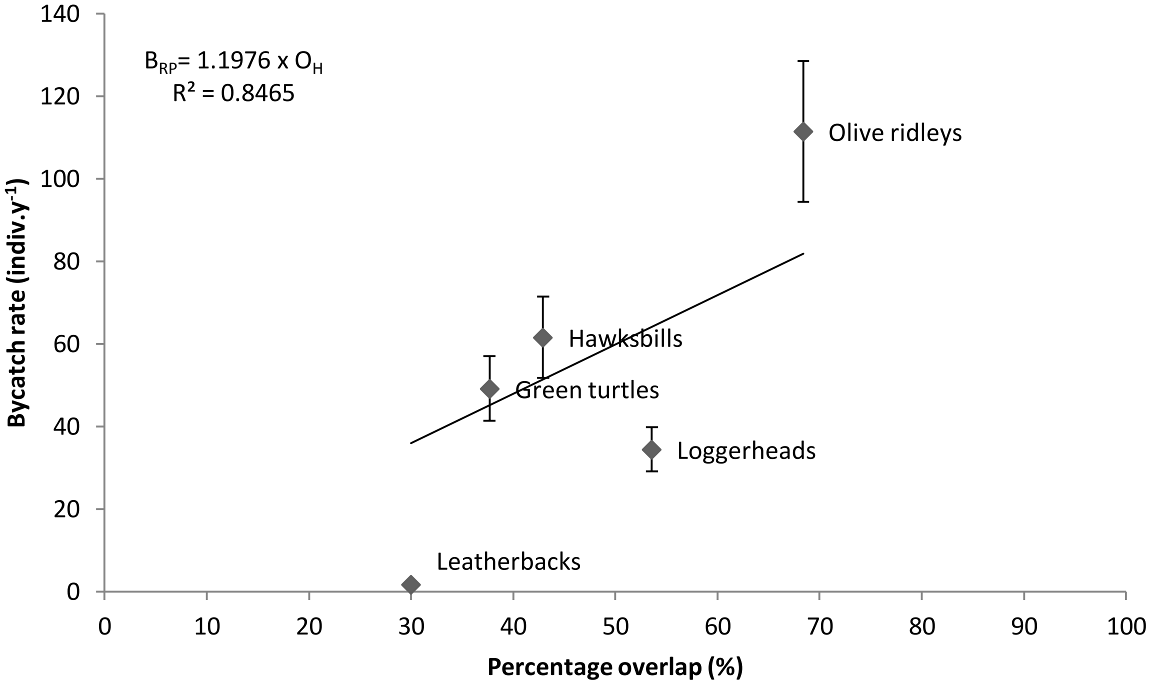


Figure 3.15 Bycatch rate (±SD, indiv.y^{-1}) per species expressed as a function of percentage horizontal overlap with the purse seine fishery in the WIO. There is a strong positive correlation between the bycatch rate and the percentage horizontal overlap, with bycatch increasing amongst species with an increase in the horizontal overlap with fisheries.

Vertical overlap

The diving depths of sea turtles differ amongst species, age classes, and habitat used (see Table 2.1 to 2.5). Maximum-recorded diving depths amongst species ranged between 91 – 1300 m, with leatherbacks diving deepest and hawksbills shallowest (Fig. 3.16, Table 3.5). The mean (±SD) diving depth from a number of studies across all age classes differed amongst species and ranged between 7.67 ± 3.30 m for hawksbills and 39.8 ± 0 m for olive ridleys (Fig. 3.17, Table 3.5).

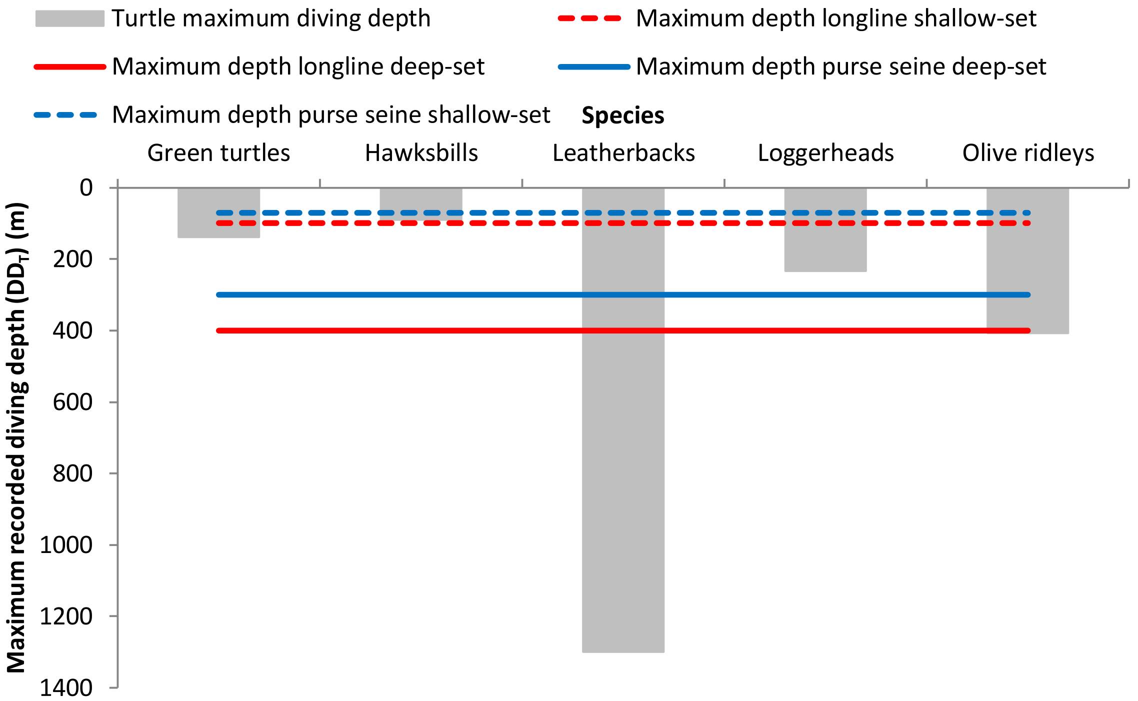


Figure 3.16 Maximum recorded diving depth ($\boldsymbol{DD_T}$) in meters for sea turtle species. Lines indicate maximum set depth for purse seine (blue; shallow-set = dashed line and deep-set = solid line), longline (red: shallow-set = dashed line and deep-set = solid line).

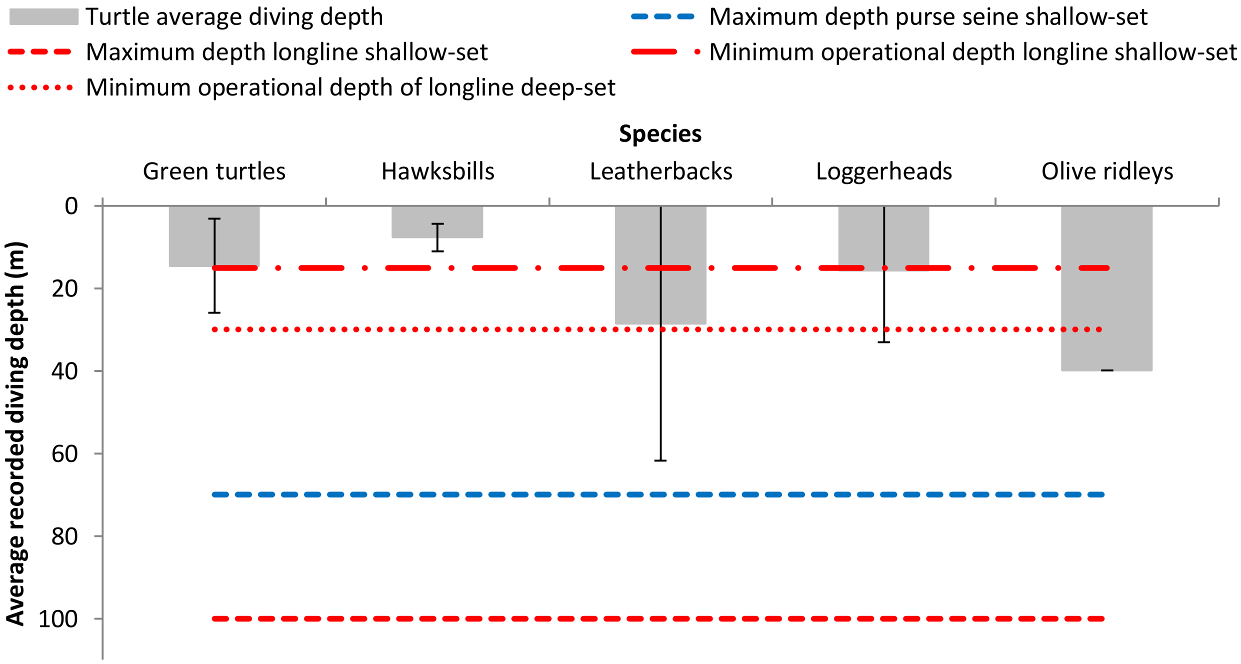


Figure 3.17 Average recorded diving depth in meters (±SD) of sea turtle species. Lines indicate maximum set depth of the shallow-set longline (red dashed line) and purse seine fishery (blue dashed line) and the minimum operational depth of the shallow-set (red dot-dash line) and deep-set longline industry (red dotted line).

Table 3.5 Maximum and average (mean ± SD) recorded diving depth for all sea turtle species. The number of studies used in the calculation of the average diving depth and the total number of sea turtles across these studies are also given.

Species	Max depth	Average depth	Number of studies	Total number of turtles
Green turtles	138	14.55 ± 11.4	16	161
Hawksbills	91	7.67 ± 3.3	9	82
Leatherbacks	1300	28.58 ± 33.08	15	148
Loggerheads	233	15.68 ± 17.36	5	20
Olive ridleys	408	39.8 ± 13.1	1	4

Maximum diving depth

There is substantial variation in the vertical overlap between deep-set longline gear (30 – 400 m depth) and the maximum diving depth of sea turtles with values ranging from 28 % for leatherbacks to 91 % for olive ridleys (Fig 3.18). Shallow-set longline gear (15 – 100 m depth) overlap ranged from 7 – 84 %, with the smallest overlap with leatherback diving and the greatest overlap with hawksbill turtles (Fig. 3.18). A Pearson product moment correlation to identify a relationship between the vertical overlap with maximum diving depths of sea turtles and bycatch rates for both the deep-set and shallow-set longline fishery indicated no correlation existed for either the shallow (r = 0.495, n = 5, p = 0.338) or deep-set longline gear deployments (r = 0.761, n = 5, p = 0.100).

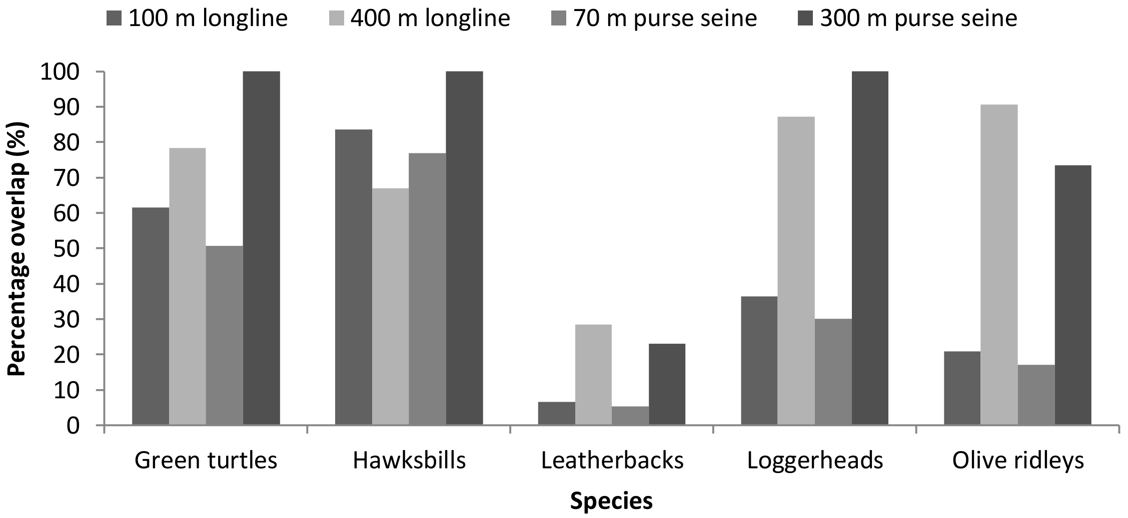


Figure 3.18 Percentage overlap between the maximum diving depths of sea turtle species and the vertical distribution of industrial longline and purse seine fisheries in the SWIO, including

overlap between deep-set and shallow-set longlines and deep-set and shallow-set purse seines.

There is complete (100 %) vertical overlap between the maximum diving depth of green turtles, hawksbills and loggerheads and the deep-set (0 – 300 m depth range) gear of the purse seine fishery, whereas olive ridleys overlapped by 74 % and leatherbacks by 23 % (Fig. 3.18). The vertical overlap between shallow-set purse seine fisheries (0 – 70 m depth range) and the maximum diving depth of sea turtle species ranged from 5 – 77 % with the least overlap with leatherbacks and the greatest overlap with loggerheads (Fig. 3.18). Pearson product moment correlations indicated no relationship between the vertical overlap and the magnitude of bycatch between species for the deep-set (r = 0.706, n = 5, p = 0.140) or for the shallow-set purse seine fishery (r = 0.474, n = 5 p = 0.360).

Average diving depth

Deep-set longline gear is set between 30 – 400 m (Beverly *et al.*, 2003). The average diving depth of most sea turtle species is shallower than 30 m except for olive ridley turtles that have an average diving depth of 39.8 m (Table 3.5). The average diving depth of olive ridley turtles thus shows a 25 % overlap with the vertical depth range of deep-set longline fishing gear (Fig 3.19). Shallow-set gear is set between 15 – 100 m depth (Beverly *et al.*, 2003). The average diving depth of sea turtles showed variable overlap with shallow-set gear, this variation depended on the species. The overlap between fishing gear and the mean diving depth for green and hawksbill turtles was 0 %, with the greatest overlap recorded for olive ridleys at 62 % overlap (Fig 3.19). Pearson product moment correlations indicated no relationship between mean vertical diving depth of sea turtles and bycatch in longline fisheries for either deep-set (r = 0.222, n = 5, p = 0.719) of shallow-set (r = 0.167, n = 5, p = 0.789) gear. For the purse seine fishery, there is complete overlap between the average diving depth of all sea turtle species and the vertical depth range of both the shallow-set and deep-set purse seine fishery.

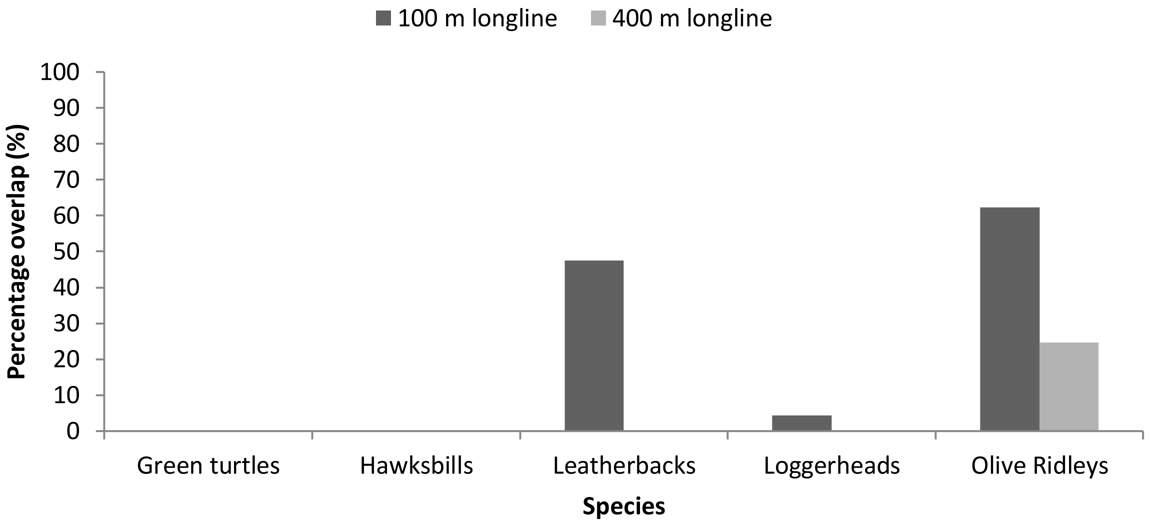


Figure 3.19 Percentage overlap between the mean diving depths of sea turtle species and the vertical distribution of industrial longline fisheries in the SWIO, including overlap between deep-set and shallow-set longline.

Discussion

The aim of this chapter was to quantify the bycatch of sea turtles in industrial longline and purse seine fisheries and to determine whether the magnitude of bycatch per species is related to the overlap with fisheries activities. The results indicated that the longline fishery has a much greater bycatch and concomitant mortality rate of sea turtles than that of the purse seine fishery. The longline fishery captures approximately 94 % of the 4388 sea turtles captured annually and purse seine only 6%. This observation is similar to other ocean regions such as the Atlantic (Angel *et al.*, 2014). The higher bycatch rates in the longline fishery compared to the purse seine fishery may be partly due to the extended distribution of the longline fishery in the region (~ 36 000 000 km^2) that is almost 3 times greater than the distribution of the purse seine fishery (~12 000 000 km^2) in the region. Other factors that may be responsible for higher capture rates in the longline fishery may include the attraction to longline bait and gear amongst others. The longline fishery also shows greater horizontal overlap with the distribution of sea turtle species (> 90 % for all species) compared to that of the purse seine fishery (30 – 70 % overlap amongst species), that allows for greater interaction with longline fishing gear and sea turtle species than with purse seine fishing gear.

The spatial extent and intensity of fishing effort varied across the region for both the longline and purse seine fisheries, although there was substantial overlap in the distribution of the two industrial fisheries within the WIO. Longline fishing takes place throughout the region with hotspots off the Horn of Africa, and in the Mozambique Channel, whereas purse seine fisheries are concentrated off Seychelles and Comoros. This variation in effort in space and time is because of piracy (Chassot *et al.*, 2010) and the variation in the distribution of the target species. The distribution of the target species are in turn determined by factors such as sea surface temperature, ocean currents, and primary productivity (Kumar *et al.*, 2014, Lehodey *et al.*, 2006). The longline effort in the WIO is about three times higher than the Eastern Indian Ocean (EIO) during the same period (IOTC, 2013a), indicating that sea turtle populations in the WIO region will face a greater risk than those in the EIO due to the greater fishing pressure. The purse seine fishery in the EIO is however only a small fraction of the effort in the WIO during the same time period (IOTC, 2013b), once again indicating that purse seine fisheries are less likely to impact on sea turtle populations in the EIO region. Over the period from 2000 – 2011 there was an initial increase in effort in both fisheries. However since 2005 and 2007 for longline and purse seine fisheries respectively there has been a decrease in effort within the region, this decrease in effort can in part be ascribed to the increase in piracy in the region within the same period of time (Chassot *et al.*, 2010).

All five species of sea turtles that occur in the region are captured in the industrial longline and purse seine fisheries in the region; however, the magnitude of bycatch varies amongst species in both fisheries. In the longline industry, loggerheads and leatherbacks are the most commonly caught species (with catch rates of 1698 and 1422 indiv.y^{-1} respectively) whereas olive ridleys, hawksbills, and green turtles are frequently caught in the purse seine industry (with catch rates of 112, 62, and 49 indiv.y^{-1} respectively). The same pattern for the longline fishery is observed in other regions where loggerheads and leatherbacks are also the most commonly caught species (Carreras *et al.*, 2004, Chaloupka & Limpus, 2001, Chan *et al.*, 1988, Lewison *et al.*, 2004b, Pinedo & Polacheck, 2004, Polovina *et al.*, 2003a). For the purse seine fishery similar patterns are observed in the Eastern (Hall & Roman, 2013) and Western Pacific (OFP, 2001). In the Atlantic Ocean Amandè *et al.* (2010) found green turtles (30 %), kemps ridleys and leatherbacks (~ 17 % for both species), to be the most commonly caught species, with olive ridleys being the least commonly caught species, whereas Clermont *et al.* (2012) reported that olive ridleys and loggerheads were most frequently caught in the Atlantic Ocean. The differences in species composition of catches can be attributed to the differences in overlap of the fisheries and the distribution of species in the various ocean basins, as well as the relative abundances of the species in the regions where the fisheries operate (Clermont *et al.*, 2012). Differences between the two datasets for the Atlantic Ocean (Amandè *et al.*, 2010, Clermont *et al.*, 2012) can be due to differences in the observer coverage between the two studies and the sizes of the datasets used.

The longline fishery has the greatest impact on the leatherback population followed by the loggerhead population. Both these species have high annual capture rates (1422 indiv.y^{-1} and 1698 indiv.y^{-1}) in the fishery. The bycatch of these species exceeds the annual number of nesting females in the region, with the bycatch of loggerhead equating approximately to double the number of nesting females, whereas for leatherbacks the bycatch equates to approximately 19.5 times the number of nesting females (number of nesting females being a proxy for population size). The capture rates of these two species are thus high relative to the population sizes, whereas the capture rates for other species are low relative to the population sizes. Despite the high levels of bycatch for the loggerhead population the annual mortality (68 indiv.y^{-1}) in the longline fishery for this species is only 7 % of the annual number of nesting females, whereas the annual mortality for leatherback turtles (57 indiv.y^{-1}) equates to approximately 79 % of the annual number of nesting leatherback females in the region.

The loggerhead population has also shown an increase in population size over the past five decades, whereas the leatherback population showed an initial increase but has shown a stable trend in the annual number of nesting females over the past two decades, despite these two species having similar protection at nesting beaches (Nel *et al.*, 2013a). It is expected that the longline fishery may be one of the factors contributing to the slow recovery rate of the leatherback population, as was found by Petersen *et al.* (2009). In other ocean regions the longline fishery is also indicated as a major risk to leatherback populations (Lewison *et al.*, 2004b, Wallace *et al.*, 2010b). There is thus greater need to mitigate the capture of leatherbacks more than any other species in longline fisheries.

The purse seine fishery has the greatest impact on the olive ridley, hawksbill, and green turtle populations in the SWIO with 112, 62, and 49 indiv.y^{1} respectively captured by this fishery. However the capture rates of these species in the fishery do not exceed the annual number of nesting females (less than 11 % for all species), and mortality rates of all species are very low when compared to the number of nesting females (all being less than 1 %). This fishery is thus not a major direct source of mortality or slowing the recovery of any of the populations in the region. The most commonly caught age class for this fishery in the WIO were juveniles (Amandé *et al.*, 2008, Clermont *et al.*, 2012), and because the growth of sea turtle populations is highly dependent on the survival of reproductively active individuals (Heppell, 1998), this further decreases the concerns regarding the impact of sea turtle bycatch in this fishery.

For both fisheries the differences in the magnitude of capture amongst species in the region is not explained by the relative abundances of sea turtle species across the region; there is no significant correlation between the abundance of nesting females per species and the magnitude of bycatch per species. For the longline fishery the differences in magnitude of bycatch is not explained by the difference in overlap either, as there is no significant correlation between the horizontal overlap (ignoring intensity) and the difference in bycatch rates amongst species. For the purse seine fishery however, there is a significant correlation between the magnitude of bycatch and the horizontal overlap with sea turtle species. The greater the horizontal overlap the greater the annual bycatch rate of a species. When considering the vertical overlap between sea turtle species and the vertical operational extent of fisheries there is no significant correlation between the magnitude of bycatch and the vertical overlap. However, vertical overlap between a species and a fishery is necessary in order for there to be the possibility of interaction between individuals of the species and the fishery.

The variation in bycatch between sea turtle species in the longline industry is related to factors such as the difference in behavioural responses to gear between the species. This includes factors such as preferred diet and attraction to gear. Species such as loggerheads and olive ridleys are more likely to be attracted by the bait than other sea turtles (Arauz, 2000, Piovano *et al.*, 2004), whereas adult green turtles, hawksbills, and leatherbacks have mainly herbivorous (Seminoff *et al.*, 2002a), spongivorous (Bjorndal, 1997) and gelatinovorous (Heaslip *et al.*, 2012) diets, respectively. Juveniles of all species are carnivorous (Frick *et al.*, 2009) or omnivorous (Bjorndal, 1997, Boyle & Limpus, 2008) when in the pelagic phase and this may lead to these juveniles being attracted to bait. Diet preference alone is not the only factor that has an impact on the magnitude of bycatch. Leatherback turtles are expected to be attracted to the light sticks used in the longline fishery (Witzell, 1999) and olive ridleys are often observed inspecting floating objects such as the float lines (Largacha *et al.*, 2005) that may increase their interaction with the fishery. The magnitude of bycatch in the longline industry is thus a function of several factors including overlap (as this is necessary for interactions), the abundance of species in a specific region (higher catches are often observed where high effort overlaps with greater abundance of species), as well as the behaviour of species that might attract them to gear.

In the purse seine fishery, the magnitude of bycatch is also not completely explained by the correlation with horizontal overlap and other factors may play a role in the differences of magnitude of bycatch. Bycatch of sea turtles in the purse seine fishery is often seen as chance encounters (Hall & Roman, 2013), however several factors can lead to increased capture rates of a species. The attraction to FADs result in higher capture rates of species in sets around these (Bourjea *et al.*, 2014). Olive ridley turtles are often seen inspecting floating objects (Largacha *et al.*, 2005) and this behaviour coupled with expected higher abundances in the fishing region with the highest effort can be said to explain the higher capture rates of olive ridley turtles in the purse seine fishery. The highest proportion of individuals captured in the purse seine fishery in the Indian Ocean was also juveniles (Amandé *et al.*, 2008, Clermont *et al.*, 2012). These juveniles are often seen in association with floating objects, and may use the FADs as a source of food or shelter. Adult individuals of most species (excluding leatherbacks and olive ridleys) display coastal distributions and thus have lower capture probabilities as fishing effort are mostly distributed away from the coast, in oceanic areas.

The mean capture rate of 0.0328 indiv.1000 hooks^{-1} in the longline industry differs from catch rates reported for other regions (Wallace *et al.*, 2010b). Higher capture rates are seen in the Northeast (0.0367 indiv.1000 hooks^{-1}), Southwest (0.224 indiv.1000 hooks^{-1}) and Northwest Atlantic Ocean (0.5954 indiv.1000 hooks^{-1}), and the Mediterranean (0.274 indiv.1000 hooks^{-1}), with lower capture rates seen in the Eastern Indian Ocean (0.019 indiv.1000 hooks^{-1}), North Pacific (0.0134 indiv.1000 hooks^{-1}), Caribbean (0.0042 indiv.1000 hooks^{-1}), and Oceania (0.0014 indiv.1000 hooks^{-1}, Wallace *et al.*, 2010b). For the WIO region a mean bycatch rate of 0.008 indiv.1000 hooks^{-1} was estimated by Wallace *et al.* (2010b), this is substantially lower than the estimate for the SWIO in this study. There are substantial differences in the bycatch rates amongst fleets within the SWIO, thus the annual impact across the fishery is highly dependent upon the effort exerted per fleet. The differences in bycatch rates may also relate to differences in the accuracy and frequency of observer coverage among fleets. The mean capture rate in the purse seine fishery in the Indian Ocean (0.03 ± 0.02 (SD) indiv.set^{-1}) is similar to that in the Atlantic Ocean (0.04 ± 0.02 (SD) indiv.set^{-1}, Bourjea *et al.*, 2014). In the Indian Ocean however there is a difference between capture on FADs and FSCs with higher capture rates around FADs than FSCs whereas in the Atlantic Ocean there is not a significant difference between the capture rates of the two sets (Bourjea *et al.*, 2014).

Even though this dissertation gives the most comprehensive study on the impacts that the industrial longline and purse seine fisheries have on sea turtle populations in the WIO, there are several limitations. Some data limitations forced the analyses to be simplistic, the estimates thus conservatively quantify the impact that these fisheries have on the sea turtle populations that nest in the SWIO region. In the absence of suitable data, the methods consequently did not take into account the multitude of factors that can have an effect on sea turtle bycatch. These factors include changes in sea turtle densities in space and time, migration routes and times, gear configuration, and setting methods (Petersen *et al.*, 2009).

Of the accessible bycatch records for the region there is often very little information other than bycatch rate and, in some cases, species-specific catch rates or percentages for species caught. The method used in this assessment was specifically focused on determining bycatch per species for specific RMUs and for the larger part ignored bycatch rates that did not offer a species delineation. Further it does not include estimates for unidentified sea turtles that are caught in fisheries. The mean bycatch rate calculated in this assessment is thus a conservative estimate. Illegal, unregulated, and unreported (IUU) fishing is a recognized international problem (Agnew *et al.*, 2009) and is likely to have an effect on bycatch estimates, increasing the impact that these types of fisheries may have on sea turtle populations (Hamann *et al.*, 2010).

An even distribution of sea turtles throughout their range was assumed due to the limited information on sea turtle densities at nesting and feeding grounds and along migration routes. This leads to the assumption that the probability of capture of sea turtles will be similar throughout the range regardless of the density of sea turtles. Capture rates per species are however expected to be higher in areas of higher sea turtle densities of a specific species where this coincides with high effort, than in areas with lower densities and the same effort. It is suggested that information that is more detailed is collected for sea turtle interactions with fisheries, including GPS coordinates, size information, and proper species identifications. This will serve to identify areas of possible bycatch hotspots, and will allow better estimates of the relative impact of the fisheries (which is a function of the size/age of the sea turtles). Detailed information regarding the at sea distribution of sea turtles will also assist in identifying areas with high sea turtle densities. This information will assist in identifying possible bycatch hotspots where high sea turtle densities overlap with high fishing effort. The identification of migration routes of sea turtle in the region will also assist in the identification of possible areas where time/area closures will be appropriate to mitigate incidental catch of sea turtles.

Differences in capture rates between the different gear configuration in both longline (shallow-set vs deep-set) and purse seine (sets on FADs vs sets on FSCs) were not taken into account as the effort data available for the region did not give clear delineation with regards to the differences of effort between the different gear configurations and sets. Better effort recording in fisheries that indicate the gear configurations and the types of sets as well as the depth of sets used per fishery will also be useful to make a better estimate of the impact of these fisheries on sea turtle populations.

Estimates of mortality in fisheries also do not include estimates of post-release mortality as no information exists for the region and estimates from other regions vary widely (Aguilar *et al.*, 1995, McCracken, 2000, NMFS, 2001a, Parker *et al.*, 2001, Swimmer *et al.*, 2002). For the purse seine fishery, the mortality associated with capture in FADs is not taken into account as not enough information exist on the number of FADs deployed in the SWIO in order to attempt to quantify the possible impact that these may have on sea turtle populations. It is expected, however, that the FADs associated with the purse seine fleet may have substantial impacts on some of the populations due to unintended mortality of sea turtles in these devices (Clermont *et al.*, 2012)

The longline industry is expected to have a demonstrable impact on sea turtle species in the SWIO region especially leatherbacks, it is thus imperative to reduce the number of sea turtles caught in this fishery. Despite the protection of sea turtles at several rookeries within the RMUs assessed, the protection at nesting beaches cannot compensate for high mortality of adults and sub-adults in fisheries. Mitigation measures (Appendix C) may lead to a decrease in bycatch rates, however if effort in the region should increase substantially it may lead to an increase in the total bycatch of sea turtles. Our understanding of the effects of fisheries remain limited due the availability of data, however it is clear that bycatch of specifically leatherback turtles should be decreased. It is however necessary to periodically reassess the impact as new information becomes available. Improved knowledge of the bycatch of sea turtles can assist in more quantitative assessments in the future. The cumulative effects of fisheries should also be taken into consideration and it thus remains a priority to decrease the bycatch of particularly those species that are at risk to fisheries such as leatherback turtles.

Chapter 4

Bycatch of sea turtles in the coastal prawn trawl, gillnet and beach seine fisheries in the SWIO

Abstract

Small-scale fisheries are important for sustainable socio-economic development of coastal communities of developing countries. However, managing these ‘informal’ fisheries is often more complicated than managing industrial fisheries. It is however important to understand the potential impacts of these activities on target and non-target species to ensure the long-term economic and environmental sustainability of these fisheries and populations. Given that sea turtle species are all listed as threatened, and that they are commonly caught in coastal fisheries, this study attempts to quantify sea turtle bycatch in coastal prawn trawl, gillnet, and beach seine fisheries in the SWIO. Data were collected from published information, online databases and technical reports to establish a database containing essential information regarding fisheries distribution and effort and sea turtle bycatch in the region. The resultant database was used to map the distribution and effort of fisheries in the region and to quantify the bycatch of each species in the respective fisheries. An estimated 52 370 indiv.y^{-1} are caught in these three fisheries. In the prawn trawl fishery a total of 1 089 – 2 795 indiv.y^{-1} are caught, with green turtles the most commonly reported as bycatch, followed by loggerheads and hawksbills. In the gillnet fishery green turtles (30 887 indiv.y^{-1}) and loggerheads (5 248 indiv.y^{-1}) are the most commonly captured species of the 40 264 indiv.y^{-1} captured in this fishery. The capture rates of all sea turtle species in the gillnet fisheries are substantial when compared to their respective population sizes. The beach seine fishery captures 9 171 indiv.y^{-1}, the greatest portion comprises green turtles (4 784 indiv.y^{-1}) followed by loggerheads (2 901 indiv.y^{-1}). The gillnet fishery is expected to be responsible for slowing the recovery rate of green turtles and leatherbacks in the SWIO region. Despite the high bycatch rate of loggerhead turtles in the gillnet fishery, the population in the SWIO region has been showing an increase in number over several decades, it is thus expected that a large number of the bycaught turtles in the gillnet fishery is from the rookeries in the northern part of the WIO region. The beach seine and prawn trawl fisheries

in isolation are not expected to slow the recovery rate of any of the populations should the effort remain at current levels. However, the cumulative effects of the coastal fisheries can be devastating to sea turtle populations. In order for these fisheries to be sustainable, the bycatch of sea turtles should be minimized in all three fisheries due to the cumulative impacts that these fisheries have.

Introduction

Small-scale fisheries play an important role in poverty alleviation and food security of coastal communities in developing countries (Garcia & Rosenberg, 2010, Kent, 1997, van der Elst *et al.*, 2005) including the SWIO region (Everett *et al.*, 2013). However, there is concern about the unintended impacts of these fisheries due to overharvesting, which include unsustainable levels of bycatch. Part of the reason for the tendency to over-exploit coastal ecosystems in which these fisheries operate is the fact that very little gear or technological impact is necessary to exploit the variety of coastal ecosystems such as coral reefs, sea grass beds, rocky shores, sandy beaches, mangroves and estuaries (Richmond, 2002).

Juvenile, sub-adult, and adult life stages of almost all sea turtle species (except for leatherbacks) spend most of their time in the neritic zone. Coastal habitats serve as important feeding, developmental and inter-nesting areas for sea turtles species (Epperly *et al.*, 1995, James *et al.*, 2005a, TEWG, 2009). However, this is also the zone where they encounter coastal fisheries particularly prawn trawl, gillnet, and beach seine operations. These fisheries have been shown to have great impacts on sea turtle populations (Kiszka, 2012b, Koch *et al.*, 2006, Moore *et al.*, 2010, Poonian *et al.*, 2008, Pusineri & Quillard, 2008). There is thus great concern regarding coastal fisheries and their interaction with sea turtles (Lewison & Crowder, 2007, Mcclellan & Read, 2007).

The artisanal nature of coastal net fisheries poses a challenge in terms of effective management strategies and reliable data collection (Panayotou, 1982, Pauly, 2006). Small-scale artisanal fisheries such as beach seine and gillnet fisheries occur primarily in developing nations (including the SWIO), and are characterized by remote landing sites that make data collection difficult (Salas et al 2007). The lack of data on these fisheries makes it difficult to assess their impacts on target species, with studies and information regarding the impact of these fisheries on sea turtle populations and other bycaught species being even more scarce (Wallace *et al.*, 2010b). There is however a need for an evaluation of the impacts of these activities on sea turtle populations, even if it is semi-quantitative, because it has been shown that the cumulative impacts of small-scale fisheries on sea turtle populations may be devastating (Moore *et al.*, 2010, Peckham *et al.*, 2007). A better understanding will allow for the direction of research and management of these fisheries to mitigate the impacts, while still taking into consideration the socio-economic concerns associated with such mitigation strategies.

Most studies in the region have focused on limited geographic distribution (e.g. single countries or specific areas within a country (Brazier *et al.*, 2012, Humber *et al.*, 2011, Pusineri & Quillard, 2008). However, it is necessary to broaden assessments to the entire distribution of the local sea turtle populations as these species occur across international boundaries and the distribution of a population can be across several countries. Therefore, the aim of this chapter is to quantify the bycatch of sea turtles in the coastal prawn trawling, gillnetting and beach seine fisheries that occur in the SWIO across the entire distribution of all sea turtle species that nest in the region. To achieve this aim, three key objectives were set. First, to quantify the extent of the fishing operations (in terms both spatial extent and effort) for prawn trawl, gillnet and beach seine fisheries. Second, to quantify the number of sea turtles caught as bycatch in each of the fisheries as well as the concomitant mortality rates. Third, to determine the vertical and horizontal overlap between fishing operations and the known distribution of sea turtles as a possible explanation for the patterns observed.

Methods

Fishery distribution

For the purpose of this study coastal fisheries were defined as those fishing operations that occur from the shoreline to 50 m from shore or those that occur at depths up to 200 m (Chuenpagdee *et al.*, 2006, Stewart *et al.*, 2010). Data on prawn trawl, gillnet, and beach seine fisheries were gathered from online databases and published reports. The variables recorded in the databases included the fishery name, fishery sector, area size, vessel type, size of vessels, number of vessels, gear type used, maximum and minimum bottom depth where fishery occurred, habitats in which the fishery operated, season and duration of the activity, and additional spatial descriptors. Data from the past ten years were used where possible. However where current information was not available, older data were used as these data are indicative of historic fishing activities and impacts (Begossi, 2006).

For fisheries extent, ArcGIS 10.2 (ESRI, 2013) was used to represent spatial distribution data where shapefiles or images the files were available. Where no shape files or images were available, the fishery boundaries were inferred from the maximum and minimum bottom depth by then relating these to bathymetry information plus any additional information available regarding fisheries boundaries (such as specific coastal spread). Where more than one type of gillnet or beach seine fishery was in operation within a country, the extreme operational depths per specific gear type were used to delimit the fishing area as effort data (e.g. number of boats) were not expressed per fishery but per gear type. If the operational bottom depth of fishery was not available, it was inferred from similar fisheries in the region with similar boats and types of fishing gear; however, where no data were available on boat type and fishing gear, the maximum depth of similar fisheries in the region was used to delimit potential fishing areas.

Boat length and number of boats are amongst the key variables used to describe fishing activity (Bordalo-Machado, 2006, Le Pape & Vigneau, 2001, McCluskey & Lewison, 2008, Piet *et al.*, 2007). Fishing effort was thus calculated as the number of boat meters.y^{-1} (F_{ET}) as well as the number of boatmeters.km^{-2} (F_{EA}). For fisheries where effort parameters were not available, only the extent of the fishery was mapped. Three basic metrics were thus extracted from the database: number of boats (N_B), length of boats (L_B in meters), area covered by the fishery (A_F in km^2). Total fishing effort (F_{ET}) and fishing effort per unit area (F_{EA}) for each of the fisheries was calculated as follows:

$$\boldsymbol{F_{ET} = N_B \times L_B} \qquad \textbf{Equation 4.1}$$

$$\boldsymbol{F_{EA} = \frac{F_{ET}}{A_F}} \qquad \textbf{Equation 4.2}$$

For determining effort in fisheries where more than one boat length was reported, average boat length was used. Where the lengths of boats were not available, the average for the type of fishery for the region was used. Where data on the number of boats were not available for the fishery, but were available for the number of fishing gears (i.e. number of gillnets) used in the fishery, the numbers of boats were inferred from the number of gears that are used per boat from similar fisheries in the region.

Bycatch calculations

Prawn trawling

Estimates for sea turtle bycatch in prawn trawling operations in the SWIO were gathered from published literature and online reports. Bycatch estimates were calculated from per-country estimates, and then added together to give an overall bycatch rate for the region per year. Recent annual bycatch rates (as indiv.y^{-1}) were available for only South Africa (2012) and Tanzania (2007); however, the Tanzanian trawl fishery is currently closed. Due to differences in reporting metrics for the other countries with prawn trawling (Kenya, Madagascar and, Mozambique), bycatch estimates were calculated per country as described below.

Bycatch rates for Kenya were available as number of sea turtles caught per trawl day and number of sea turtles caught per year (B_{YT}). No information was available on the number of trawl days per year for the fishery, thus it was opted to calculate a bycatch rate (B_R, indiv.vessel^{-1}.y^{-1}) per vessel per year using the number of active vessels in 1992 that corresponds to the period of the bycatch record (N_{HV}) in the fishery as follows:

$$\boldsymbol{B_R = \frac{B_{YT}}{N_{HV}}} \qquad \textbf{Equation 4.3}$$

The resultant bycatch rate (B_R, indiv.vessel^{-1}.y^{-1}) was then used to calculate an annual bycatch rate (B_T, indiv.y^{-1}) using the number of active vessels in 2012 (N_{AV}) to scale the bycatch to current effort levels using the following equation:

$$\boldsymbol{B_T = B_R \times N_{AV}} \qquad \textbf{Equation 4.4}$$

No information was available regarding a mortality rate of bycaught sea turtles in the fishery in Kenya.

Bycatch rates for Madagascar were available as the number of sea turtles per year. As for Kenya bycatch rates (B_R, indiv.vessel^{-1}.y^{-1}) and annual bycatch rate (B_T, indiv.y^{-1}) was calculated using equations 4.3 and 4.4. An average mortality rate ($M_{\%}$) was used to calculate the total annual mortality (M_T, indiv.y^{-1}) using the following equation:

$$\boldsymbol{M_T = B_T \times M_{\%}} \qquad \textbf{Equation 4.5}$$

For Mozambique bycatch rates were available per vessels per month for the summer (B_{RS}, inidv.vessel^{-1}.month^{-1}) and winter (B_{RW}) seasons. According to Gove *et al.* (2001) "summer" lasts for 7 months and winter months span the rest of the year. Annual bycatch (B_T, indiv.y^{-1}) was calculated as follows:

$$\boldsymbol{B_T = (B_{RW} \times N_M \times N_V) + (B_{RS} \times N_M \times N_V)} \qquad \textbf{Equation 4.6}$$

where N_M is the number of months per season and N_V is the number of active vessels.

No information regarding mortality was available for the Mozambican prawn trawl fishery.

Gillnetting and beach seining

Bycatch rates for sea turtles in gillnet and beach seine operations in the SWIO were gathered from published literature and online reports. Bycatch estimates were calculated from per-country estimates, and then added together to give an overall annual bycatch estimate for the region (indiv.y^{-1}). Recent annual bycatch rates were available for South Africa and Mauritius. All other countries where gillnetting and beach seining take place were treated separately, with calculations done as described below.

Comoros

There were not enough available data regarding the bycatch of sea turtles in the beach seine fishery of the Comoros in order to calculate an annual bycatch rate for this fishery. For the gillnet fishery the following information was available: percentage of fishers (boats) that reported capturing sea turtles ($F_{\%}$) and number of boats used in the fishery (N_V). The number of boats that are reported to capture sea turtles (N_{VT}) was calculated as follows:

$$\boldsymbol{N_{VT} = F_{\%} \times N_V} \qquad \textbf{Equation 4.7}$$

The average number of sea turtles (N_T) that are caught per fisher (boat) annually (indiv.y^{-1}) for those fishers that reported captures were available. Using this information annual bycatch rate (B_T, indiv.y^{-1}) was calculated for the gillnet fishery as follows:

$$\boldsymbol{B_T = N_{VT} \times N_T} \qquad \textbf{Equation 4.8}$$

Mortality per year was calculated using Equation 4.5.

Kenya, Mozambique and Tanzania

Bycatch rates per vessel per year were available (B_R, indiv.vessel.y^{-1}) for these countries as well as the number of vessels (N_V) per fishery. Annual bycatch rates (B_T, indiv.y^{-1}) were calculated as follows:

$$\boldsymbol{B_T = B_R \times N_V} \qquad \textbf{Equation 4.9}$$

Mortality per year was calculated using Equation 4.4.

Madagascar

The available bycatch rate for Madagascar was expressed as the number of individuals caught annually per kilometer of coastline (indiv.km^{-1}.y^{-1}). The bycatch rate was for landed sea turtles, so it is thus assumed that there is a 100 % mortality rate. The annual bycatch rate (B_T, indiv.y^{-1}) and annual mortality rate (M_T, indiv.y^{-1}) were calculated as follows:

$$\boldsymbol{M_T = B_T = B_R \times L_C} \qquad \textbf{Equation 4.10}$$

where L_C is the total length (km) of the Madagascar coastline (ASCLME/SWIOFP, 2012).

Mayotte

For Mayotte, the following information was available: the total number of annual sea turtle mortalities in artisanal fisheries (M_A, indiv.y^{-1}), the percentage that can be ascribed to net fisheries ($N_{\%}$) (gillnet and beach seines), and the percentage mortality for bycaught sea turtles ($M_{\%}$). Total annual bycatch (B_T, indiv.y^{-1}) and mortality (M_T, indiv.y^{-1}) was calculated using the following equations:

$$\boldsymbol{M_T = M_A \times N_{\%}} \qquad \textbf{Equation 4.11}$$

$$\boldsymbol{B_T = \frac{M_T}{M_{\%}}} \qquad \textbf{Equation 4.12}$$

Overlap between fisheries and sea turtles

The horizontal and vertical overlap between fisheries and sea turtle species were calculated as per methods in Chapter 3. The vertical overlap calculations were done for the operational depths at which the trawl (10 – 70 m), gillnet (0 – 200 m) and beach seine (0 – 30 m) fisheries occur in the SWIO. Horizontal and vertical overlap values were obtained for all five species for all three of the fisheries.

To determine whether bycatch per species was related to the degree of overlap the bycatch data was tested for normality using the Shapiro Wilk test (Shapiro & Wilk, 1965). For data that fit the assumption of normality, Pearson product moment correlations (Pearson, 1901) were performed to assess the correlation between overlap and the magnitude of bycatch. Spearman rank correlations (Spearman, 1904) were done where normality of data was not a reasonable assumption. All calculations were performed in Excel and statistical analyses in R (R Core Team, 2014).

Results

Fishery distribution

Trawling

Countries where prawn trawl fisheries were operational within the past decade in the SWIO include Kenya, Madagascar, Mozambique, South Africa, and Tanzania (Fig. 4.1 and Table 4.1), however the Tanzanian prawn trawl fishery has been closed since 2008 (Everett *et al.*, 2013). In Kenya the trawl fishery was closed in 2002 and reopened in 2011 with only 2 active vessels trawling in 2013 (WIOFISH, 2011). Prawn trawl fisheries that are currently active in the SWIO span across 65 521 km^2 of the region (Table 4.1). There is great variation in effort across the region with each of the fisheries in the region having a closed season (Table 4.1). Fishing effort varied among countries and ranged 35 – 1 910 boat meters.y^{-1}, with effort per km^2 of fishing area ranging between 0.038 – 0.105 boat meters.km^{-2}.y^{-1} (Table 4.2). The largest total effort was in Mozambique (1 910 boat meters.y^{-1}) and Madagascar (714.5 boat meters.y^{-1}), with the greatest effort per km^2 in Madagascar (0.105 boat meters.km^{-2}.y^{-1}) and South Africa (0.053 boat meters.km^{-2}.y^{-1}).

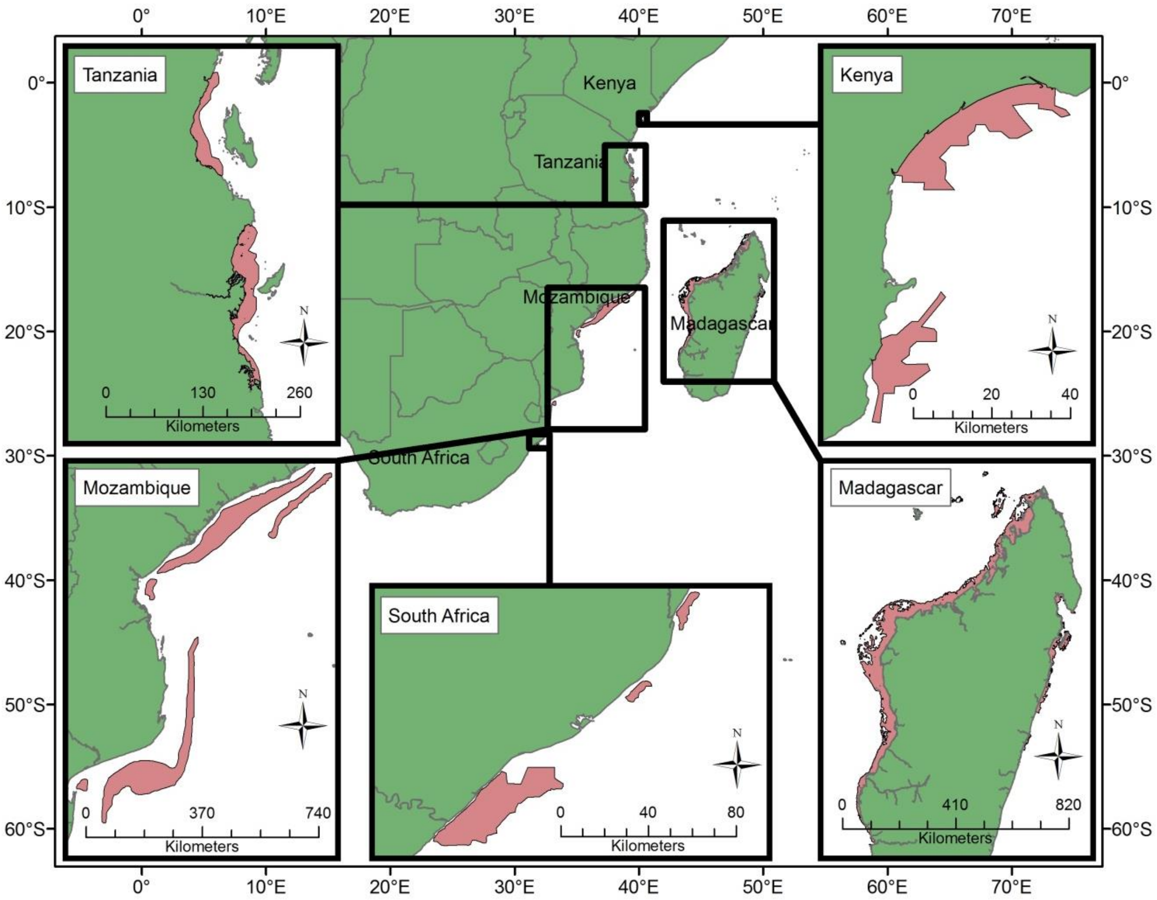


Figure 4.1 Extent of prawn trawl fishing grounds in the SWIO (following Fennessy (2012), Fulanda *et al.* (2011), Munga *et al.* (2012), Silas (2011), and Razafindrainibe (2010). Trawling grounds in Tanzania are currently closed.

Table 4.1 Summary of the most recently available data for prawn trawl fisheries in the SWIO for the period between 2000 - 2013. Details of fisheries include: fisheries sector (I = Industrial, A = Artisanal, SI = Semi-Industrial, and SSC = Small-Scale Commercial), size of fishing grounds (km^2), the type of vessels used (B = Beam trawler, T = Trawler, M = Mini trawler, P = Prawn trawler, and S = Stern trawler), the length of vessels (m), the number of active vessels, gear used (O = Otter trawls, BE = Beam trawls, BO = Bottom trawls, and S = Shallow shrimp trawls), minimum (min), and maximum (max) bottom depth, and habitats of operation (GIA = General inshore area, SCB = Shallow coastal bays, IZ = Intertidal zone, SS = Sandy seabed, SB = Sea grass beds, OB = Oceanic bottom, and OM = Oceanic mid-water).

Country	Sector	Area (km^2)	Vessel type	Vessel length (m)	No. active vessels	Gear Type	Min Depth (m)	Max Depth (m)	Habitat	Season	Refs[*]
Kenya	I	1202	B	25	2	O	12	70	GIA SCB	Apr – Nov	1, 2, 3, 4, 5
Madagascar	I A	6800	T M	17 8-11	37 9	O	2	30	GIA IZ SCB	Mar - Nov	2, 3, 6
Mozambique	I	50000	P B	30 13	44 26	BE BO S	5	70	GIA SS SB SCB	Mar – Sep (Sofala Banks) Mar – Dec (Maputo Bay)	2, 3, 7
	SI		P	12	21						
South Africa	I	650	S	35	1	O	10	50	GIA	Mar - Aug	2, 3, 8
Tanzania	I SI SSC	6869	P	24	25	S	5	20	GIA OB OM SB	Fishery closed in 2008	2, 3

[*]
1. KDF (2012)
2. WIOFISH (2011)
3. Fennessy (2012)
4. Fulanda *et al.* (2011)
5. Munga *et al.* (2012)
6. Razafindrainibe (2010)
7. Gove *et al.* (2001)
8. Nel *et al.* (2013a)

Table 4.2 Total annual fishing effort (boat meters.y^{-1}) and annual effort per unit area (boat meters.km^{-2}) in prawn trawl fisheries in the SWIO that were actively operating between 2000-2014. Records for the Tanzanian fishery that is currently closed are not included.

Country	**Total effort (boat meters)**	**Effort per unit area (boat meters.km^{-2})**
Kenya	50	0.042
Madagascar	714.5	0.105
Mozambique	1910	0.038
South Africa	35	0.053

Gillnetting

In the SWIO region, gillnet fisheries are operational in Comoros, Kenya, Madagascar, Mauritius, Mayotte, Mozambique, Seychelles, South Africa, and Tanzania (Table 4.3, Fig. 4.2, and Fig. 4.3), with the total area covered by the gillnet fishery amounting to 300 197 km^2 of the SWIO region (Table 4.3). There is substantial variation in methods and operations across the region, but most of the fisheries operated throughout the year with very few fisheries having closed seasons. Fishing effort depended on the country with total effort per country ranging from the lowest for Mauritius at 12 boat meters.y^{-1}, to the highest for Mozambique at 44 209 boat meters.y^{-1}, and annual effort per unit area varying between the lowest for Seychelles at 0.003 boat meters.km^{-2}.y^{-1} and the highest for Comoros at 3.76 boat meters.km^{-2}.y^{-1} (Table 4.4).

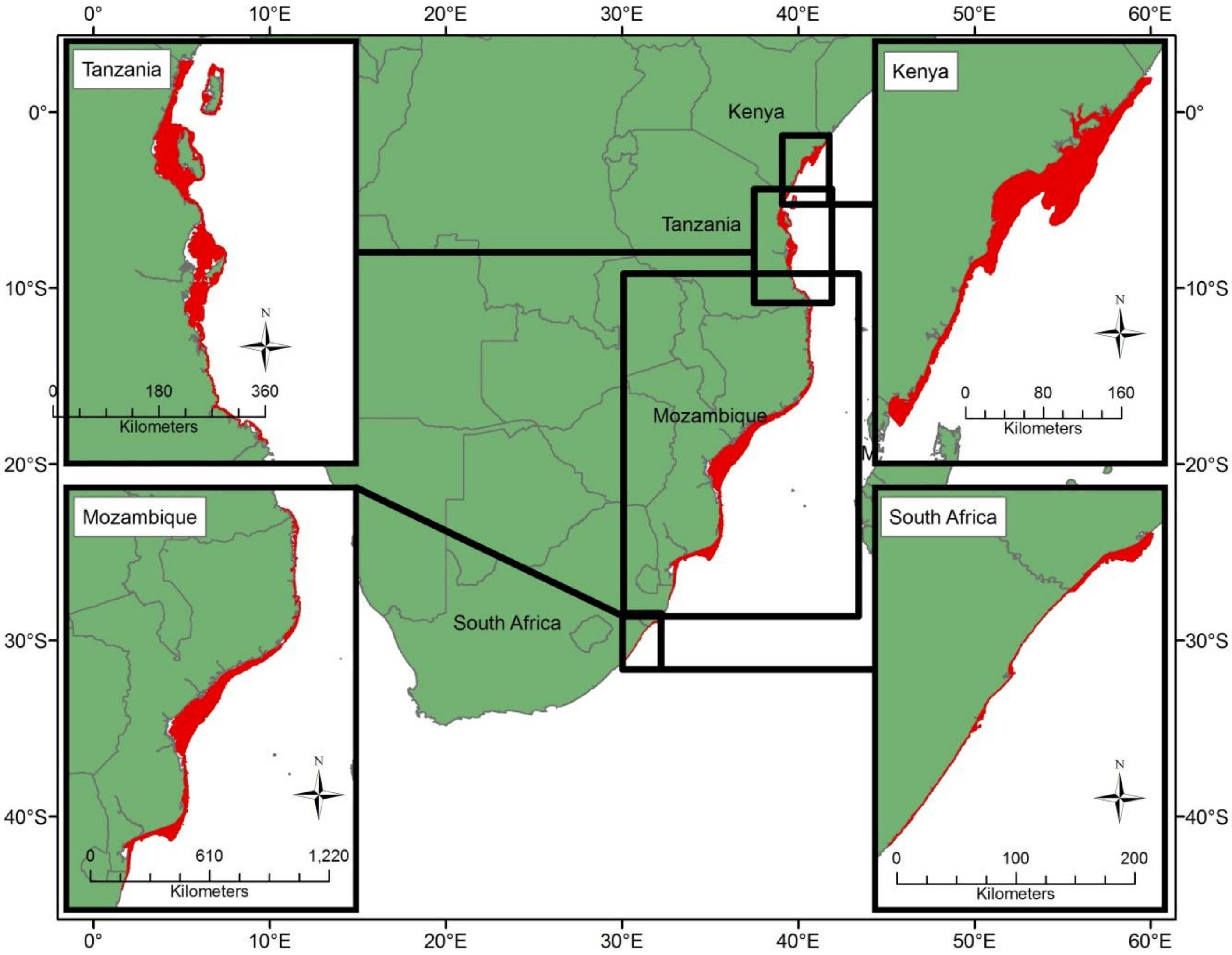


Figure 4.2 The extent of gillnet fishing areas of continental countries in the SWIO shown in red.

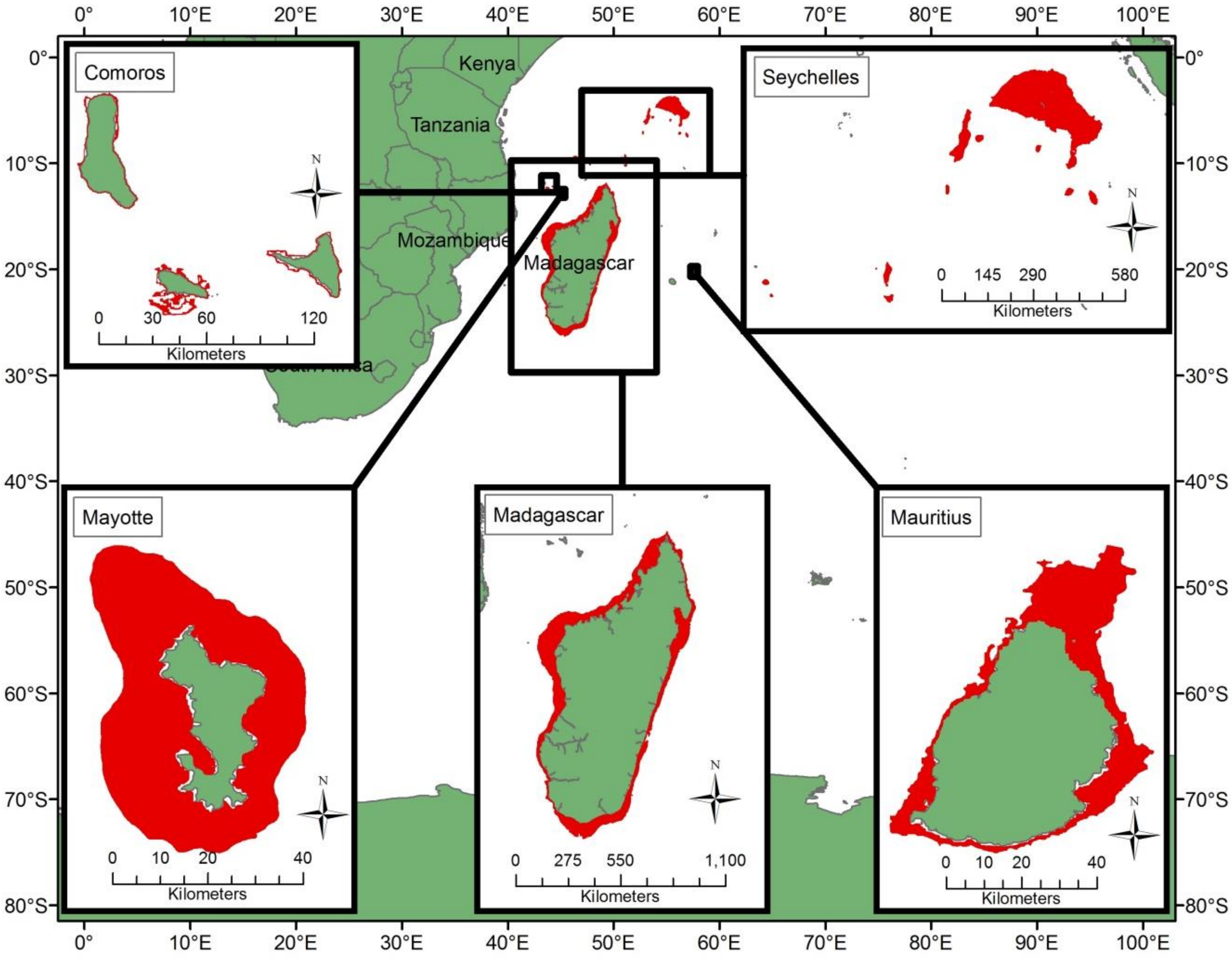


Figure 4.3 The extent of gillnet fishing areas of island nations in the SWIO shown in red.

Table 4.3 Summary of gillnet fisheries in the SWIO. Details per fishery include: fishery type, fishery sector (A = Artisanal, SC = Semi-commercial, S = Subsistence, SI = Semi-industrial, B = Bather protection), area (km^2), vessel type (B = Boti, C = Chata, D = Dau, G = Gillnetter, K = Kwassa Kwassa, M = Mashua, MC = Moma Canon, MM = Mini Mahe, N = Ngalawa, P= Pirogue, S = Ski boat, T = Trunk canon, V = Vedette), length (L) of vessels (m), number of vessels, gear used (B = Bottom-set gillnet, C = Chicocota, D = Driftnets, F = Fixed gillnets, G = Gillnets, S = Surface gillnets, SG = Surrounding gillnets), minimum (min), and maximum (max) bottom depth (m), habitats of operation (CPR = Carpet/Patch reef, CRP = Coral reef platforms, CRS = Coral reef slopes, D = Drop offs, E = Estuaries, GIA = General inshore area, GOA = General offshore area, GRS = Granite reef slopes, IMF = Intertidal mud flats, ISF = Intertidal sand flats, IZ = Intertidal zone, L = Lagoons, MB = Macro algal beds, MC = Mangrove creek, OB = Oceanic bottom, OS = Oceanic surface, RS = Rocky seabed, RUS = Rubble seabed, SB = Sea grass bed, SCB = Shallow coastal bays, SS = Sandy seabed), and operational seasons.

Country	Type	Sector	Area (km^2)	Vessel type	Vessel length (m)	No. active vessels	Gear type	Min depth (m)	Max depth (m)	Habitat	Season	Refs*
Comoros	Drift net	A SC S	229	P	3	191	D	6	18	GIA, OS, MB, GIA, CRS	Sep – May	1 - 8
	Bottom set			P	3		F	3	12		Year round	
				K	6							
	Surface			K	6		S	3	18		Sep – May	
	Surrounding net			K	6		SG	3	12		Sep – May	
Kenya	Bottom set	A SC S	8837	N	10	**1225**	B	0	10	CRS, GIA, E, GOA, IZ, MC, OS, RUS	Year round#	1 - 3, 9 - 11
				B	10							
	Crustacean			B	4		G	0	10		Year round	
	Surface			N	8		S	0	20		Year round#	
				M	4		F					
Madagascar	Small gillnet	A SC S	135000	V	9	100	G	5	200	CRS, DO, GIA, IMF, GOA, OB, OM, OS, RS, E, IZ, MC, SCB, CPR, CRP, ISF, MB L, SB	Year round	1 - 3
				P	7	30		20	200			
	Small gillnet			P	3		G	0	50		Mar – Nov	
				P	6	5000						
	Surface gillnet			P	5	200	G	5	50		Year round	
				V	9	30						
Mauritius	Fixed gillnet	A	1217	P	6	2	F	0	3	L	Mar- Sep	1 - 3
Mayotte	Set gillnet	A	1768									1 - 3

Table 4.3 Continued

Country	Type	Sector	Area (km^2)	Vessel type	Vessel length (m)	No. active vessels	Gear type	Min depth (m)	Max depth (m)	Habitat	Season	Refs*
Mozambique	Bottom set	A	71015	T	4	3528	B	10	15	GOA, GIA, IMF, ISF, E, IZ, SS, SB, SCB	Year round	1 – 3, 12
	Chicocota net			MC C	5 6	1287 125	C	1	15		Year round	
	Crustacean			C	4		B				Mar – Dec	
	Small pelagic fish			MC C	4 4	3372 2356	F	1	100		Year round	
	Sharks, rays and fish			C	7		S	5	20		Year round	
Seychelles	Bottom set	A	67100	MM	6		B			SCB, CRS, GIA, GRS		1 – 3, 13 - 15
	Sharks					28	F	0	10			
	Mackerel			MM	6		S				Year round	
Tanzania	Bottom-set	A SI S SC	14324	D B	7 9	628 810	B	6	10	CRS,SB, GIA, RS, GOA, OS, CRP, RUS, L, MB, OM	Year round	1 - 3
	Surface			M D	10 8	650 650	S	5	100		Year round	
	Surrounding gillnet			B	12	750	G	10	30		Year round	
South Africa	Surface	B	707	S	5	15	F	10	14	GIA	Year round	1

[#] Effort decreases during the southeast monsoon

*

1. WIOFISH (2011)
2. Kiszka and Muir (2007)
3. Kiszka (2009)
4. Williams James (1988)
5. FAO (2003-2015)
6. Poonian *et al.* (2008)
7. Abdoulhalik (1998)
8. Union des Comores (2005)
9. FAO (2007-2015)
10. KDF (2012)
11. Maina (2012)
12. Menezes (2008)
13. Lablache *et al.* (1988)
14. Payet (1996)
15. SFA (2009)

Table 4.4 Total annual fishing effort (boat meters.y^{-1}) and annual effort per unit area (boat meters.km^{-2}) in active gillnet fisheries in the SWIO. Fisheries for which data was not available were listed as data deficient (DD).

Country	**Total effort (boat meters)**	**Effort per unit area (boat meters.km^{-2})**
Comoros	860	3.755
Kenya	8983	1.017
Madagascar	32380	0.240
Mauritius	12	0.010
Mayotte	DD	DD
Mozambique	44209	0.623
South Africa	75	0.106
Seychelles	168	0.003
Tanzania	32386	2.261

Beach seining

Beach seine fishing took place in Comoros, Kenya, Madagascar, Mauritius, Mayotte, Mozambique, Reunion, Seychelles, South Africa, and Tanzania (Table 4.5, Fig. 4.6 and 4.7). The beach seine fishing areas cover 60 529 km^2 of the SWIO region (Table 4.5). Fishing effort varied among countries although very few had closed seasons in their fisheries, even though it may not be operational year round. Total effort ranged from lowest for Seychelles (5 boat meters.y^{-1}) to highest for Comoros (13 200 boat meters.y^{-1}). When annual effort was expressed per unit area (which is more indicative of fishing pressure), Mozambique ranked lowest at 0.19 boat meters.km^{-2}.y^{-1} and Comoros highest at 18.91 boat meters.km^{-2}.y^{-1} (Table 4.6).

Table 4.5 Summary of beach seine fisheries in the SWIO. Details on fishery type, fishery sector (A = Artisanal, S = Subsistence, SSC = Small-scale commercial), area size where beach seine fishery takes place (in km^2), vessel type (K = Kwassa kwassa, P = Pirogue, B = Boti, N = Ngalawa, M = Mashua, D = Dau, MT = Mtumbwi, SD = Seine net dory, S = Seiner, I = Inflatable vessel), length of vessels (in meters), number of vessels, the gear type used (S = seine nets, R = reef seine, B = beach seine), the minimum (min) and maximum (max) bottom depth at which the fisheries operate, the habitats in which the fisheries operate, and the operational season. For fisheries where data was not available, these were labelled as data deficient (DD).

Country	Type	Sector	Area (km^2)	Vessel type	Vessel length (m)	No. active vessels	Gear type	Min depth (m)	Max depth (m)	Season	Habitat	Refs*
Comoros	Beach seine	A SSC	698	K P	6 3	3300	S	0	5	Year round	IMF, ISF, IZ, GIA	1 - 8
	Fish herding	S		P	3		S	5	30	Sep - May		
Kenya	Reef seine	A S	2447	B N	4 10	261	R	4	10	Year round	CRP, GIA	1 – 3, 9 - 11
	Beach seine			M D	4 3		B	0	6	Year round	MB, SB, SEB, SCB, E, GIA, MC, SCB	
	Crustacean			D MT	3 4		S	0	6	Year round		
Madagascar	Beach seine	A,S	7931	P	3	DD	B	1	5	Year round	SB	1 - 3
Mauritius	Beach seine	A	158	P	7	24	S	1	10	Mar-Sep	GIA, L, SCB	1 - 3
Mayotte	Beach seine		1180			DD						1 - 3
Mozambique	Beach seine	A	37295	SD	3	2384	S	0	20	Feb – Dec	SB	1 – 3, 12
Reunion	Beach seine		162			27						13
Seychelles	Beach seine	A	2	P	5	1	B	0	20	Year round	SCB	1 – 3, 14 - 16
South Africa	Beach seine	A	709	S	6	3	B	0	20	Year round	IZ	1
	Sardines			I		28	B	0	20	May – Jul		

Table 4.5 Continued

Country	Type	Sector	Area (km^2)	Vessel type	Vessel length (m)	No. active vessels	Gear type	Min depth (m)	Max depth (m)	Season	Habitat	Refs*
Tanzania	Beach seine	A	9947	MT	4	768	B	1	5	Year round	E, GIA, IMF, ISF, IZ, L , MB, MC, SB, SSB, SEB, SCB, CRB, CRP, CRS,	1 - 3
	Boat drag	S		B	8	1171	S	2	20			

1. WIOFISH (2011)
2. Kiszka and Muir (2007)
3. Kiszka (2009)
4. Williams James (1988)
5. FAO (2003-2015)
6. Poonian *et al.* (2008)
7. Abdoulhalik (1998)
8. Union des Comores (2005)
9. FAO (2007-2015)
10. KDF (2012)
11. Maina (2012)
12. Menezes (2008)
13. Kiszka (2009)
14. Lablache *et al.* (1988)
15. Payet (1996)
16. SFA (2009)

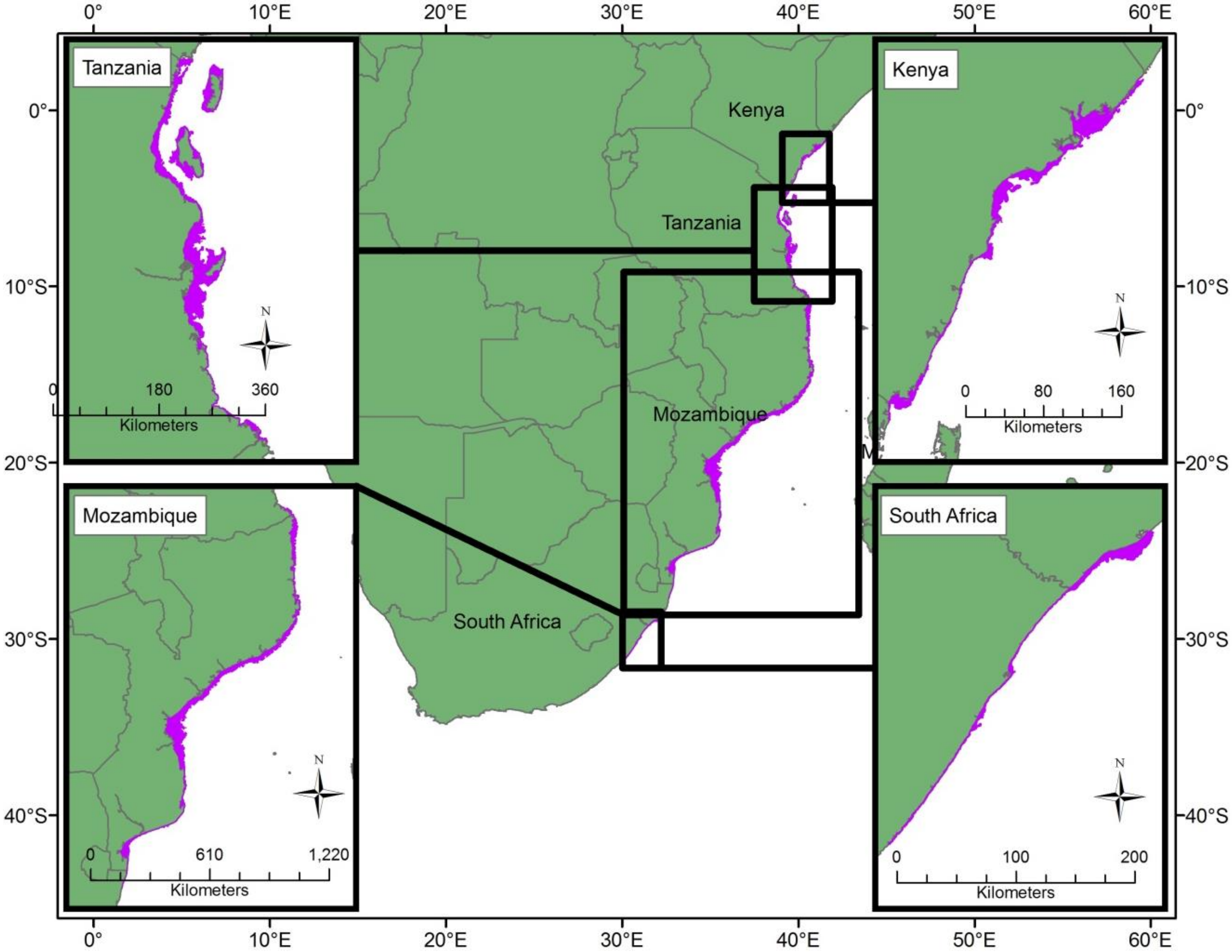


Figure 4.4 Extent of beach seine fishing areas of continental countries in the SWIO shown in purple.

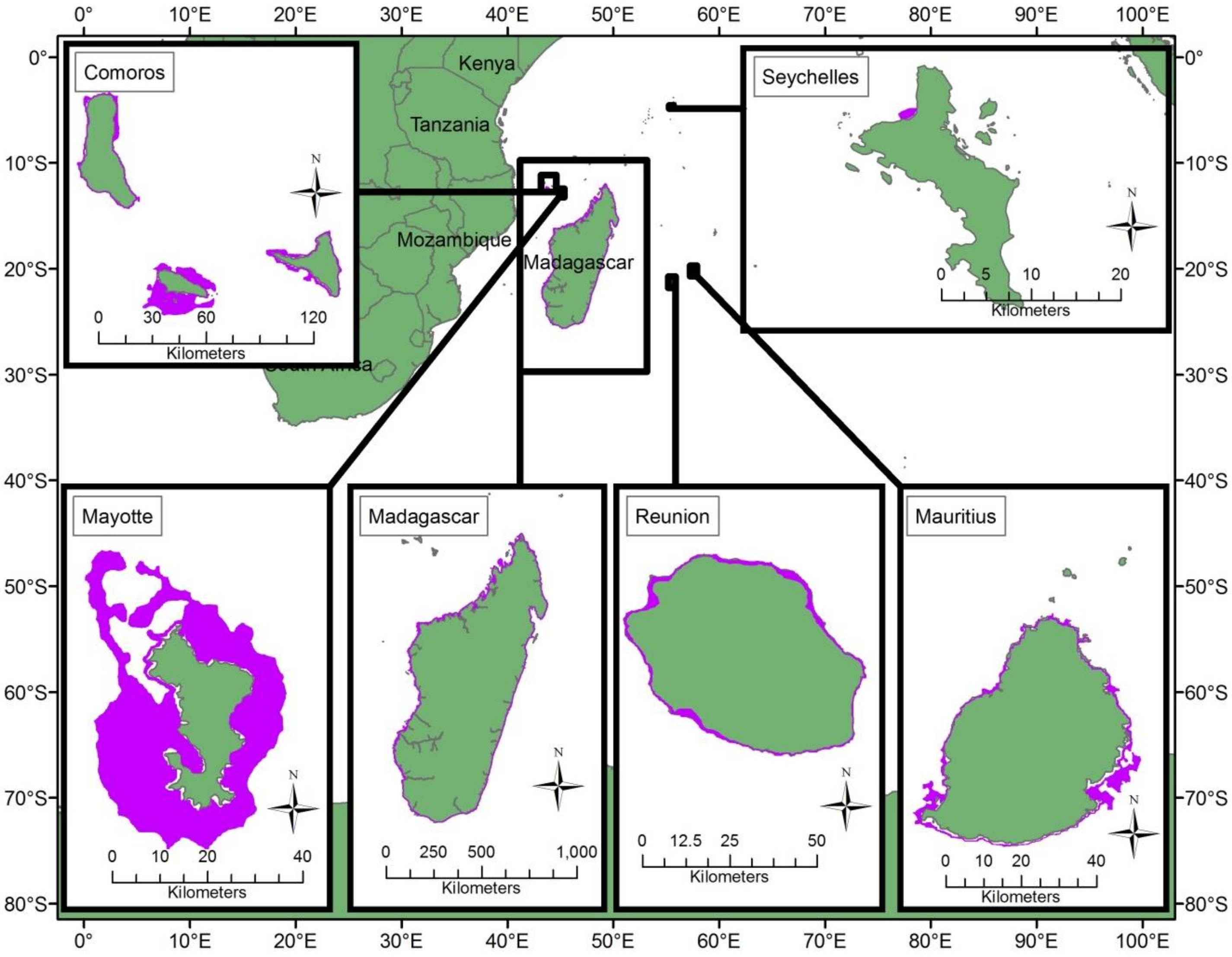


Figure 4.5 Extent of beach seine fishing areas of island nations in the SWIO shown in purple.

Table 4.6 Total annual fishing effort (boat meters.y^{-1}) and annual effort per unit area (boat meters.km^{-2}) in active beach seine fisheries in the SWIO. (DD = Data defficient).

Country	**Total effort (boat meters.y^{-1})**	**Effort per unit area (boat meters.km^{-2})**
Comoros	13200	18.91
Kenya	1216	0.50
Madagascar	DD	
Mauritius	168	1.06
Mayotte	DD	
Mozambique	7152	0.19
Reunion	DD	
Seychelles	5	2.50
South Africa	186	0.26
Tanzania	12440	1.25

Bycatch estimates and estimated mortality at the time of capture

Trawling

Bycatch rates were available for each of the countries where prawn trawl fisheries occurred (Table 4.7 and references therein). An estimated 1 089 – 2 795 indiv.y^{-1} were caught in active prawn trawl fisheries in the SWIO, with the per country estimates ranging between 2 - 184 indiv.y^{-1} caught in Madagascar to an estimated 1 008 – 2 394 indiv.y^{-1} caught in Mozambique (Table 4.7). It was not possible to estimate species composition of catches but green turtles were (by far) the most commonly captured species, followed by loggerheads and hawksbills. Leatherbacks and olive ridleys were seldom reported as bycatch throughout the region.

No estimate of mortality in the fishery was possible as mortality rates were available for only South Africa and Madagascar. Mortality rates are also reasonably specific to an operation and dependent on gear types and tow times. Mortality rates are thus not applicable among regions or ocean basins. For example, even within-country, e.g. the Australian prawn trawl fishery the mortality rate of sea turtles varied from 1 % (Robins & Mayer, 1998) to 10 – 18 % (Poiner & Harris, 1996) with the percentage of comatose sea turtles varying between 3 – 4 % (Robins & Mayer, 1998) and 21 % (Poiner & Harris, 1996).

Table 4. 7 Summary of bycatch rates in the prawn trawl fisheries in the SWIO per country, including bycatch details, species caught, number of vessels, the bycatch rate and the estimated annual bycatch rate.

Country	Bycatch details	Species	Vessels	Bycatch rate	Annual rate (indiv.y^{-1})
Kenya	2-3 turtles per day that trawling occurs[1] 500-1000 turtles per year before the use of TEDs[2]	Mostly green turtles[3]	17 active in 1992[4] 12 active in 1996[3]	From available information a bycatch rate of between 29.41 - 83.33 turtles per boat per year is calculated	59 – 167
Madagascar	5 vessels on east coast caught 20 turtles[5] (rate of 4 indiv.vessel^{-1}.y^{-1}) 2004 - 2010 : 2 – 120 turtles per year, mortality rate of between 0 - 14 %[5]	All 5 species are represented however no catch composition is available[5]	46 active vessels[6]	From available information it is estimated that between 2 to 184 turtles are caught per year	2-184
Mozambique	Bycatch rate for winter of 0.8-2 indiv.month^{-1}.vessel^{-1}, summer rate of 4-8 indiv.month^{-1}.vessel^{-1} for semi-industrial fleet. Industrial vessels have 2x the bycatch rate of semi-industrial[7]	Green turtles and leatherbacks recorded but no specific species composition available[7]	91 active vessels[6]	Semi-industrial: winter rate of 0.8-2 turtles.month^{-1}.vessel^{-1} and summer rate of 4-8 turtles.month^{-1}.vessel^{-1} Industrial vessels are double that of semi-industrial	1008 – 2394
South Africa	20 – 50 mortalities per year[8] 0.13 loggerheads caught per trawl[9] 0.0008 leatherbacks caught per trawl[9]	Mostly green turtles and loggerheads[8]	1 active vessel[4]	20 – 50 mortalities per year	20-50
Tanzania	54 turtles per year (69 % adults and sub adults)[10,11] 76.4 turtles per year[12]	62.5 % green turtles, 19 % hawksbills,12.5 % loggerheads, 6 % unidentified[12]	Fishery closed 0 active vessels[6]	54 – 76.4 indiv.y^{-1}	54-76

References

1. Mueni and Mwangi (2001)
2. Wamukoya and Mbendo (1995) in Okemwa *et al.* (2004a)
3. Wamukoya (1996)
4. Fennessy (2012)
5. Razafindrainibe (2010)
6. WIOFISH (2011)
7. Gove *et al.* (2001)
8. Van der Elst (2012)
9. (Nel *et al.*, 2013a)
10. Joynson-Hicks and Ngatunga (2009)
11. West (2010)
12. Muir (2005)

Gillnetting and beach seine fisheries

Capture rates varied among fisheries, species, and countries. Coastal gillnet and beach seine fisheries operate in several countries across the region, with bycatch data available for each of the countries (Table 4. 8). Overall, sea turtle bycatch in the SWIO coastal gillnet and beach seine fisheries combined was estimated at 49 575 indiv.y^{-1}, with an associated mortality of 34 555 indiv.y^{-1} (69.7 %) Gillnet fisheries were responsible for approximately 81 % of this catch, totalling 40 264 indiv.y^{-1}, with a mortality rate of 29 964 indiv.y^{-1} (Table 4.9). Beach seine fisheries in the SWIO region captured 9 171 indiv.y^{-1} with 49.5 % mortality (4 544 indiv.y^{-1}, Table 4.9). These estimates excluded catches from Mayotte because the data from Mayotte combined catches for these two fisheries (bycatch = 140 indiv.y^{-1}; mortality = 46 indiv.y^{-1}).

Per species capture rates for gillnet fisheries ranged 86 – 30 887 indiv.y^{-1} with the highest capture rates recorded for green turtles (30 887 indiv.y^{-1}) and second highest for loggerhead turtles (5 248 indiv.y^{-1}). The lowest catch estimate (86 indiv.y^{-1}) was for leatherbacks (Table 4.9 and Fig 4.6). The number of individuals per species caught expressed as a percentage of the number of nesting females varied among species. It was lowest for hawksbills at 91.24 % and highest for loggerhead turtles at 535.51 % (Table 4.10). A Shapiro-Wilk test indicated that normality is not a reasonable assumption for the gillnet bycatch data (W = 6861, p = 0.007). A Spearman rank correlation was thus run and a significant strong positive correlation exists between the two variables (Spearman's rho = 0.7, D_f = 3., P = 0.23), i.e. bycatch rate and abundance of species.

Mortality rates per species for the gillnet fishery ranged from the lowest for leatherbacks (30 indiv.y^{-1}) to the highest for green turtles (27 272 indiv.y^{-1}; Table 4.10). The number of mortalities per species expressed as a percentage of the number of nesting females also varied amongst species with the lowest for olive ridleys (20.38 %) and the highest for green turtles (257.31 %; Table 4.10).

Table 4.8 Summary of bycatch rates per country of gillnet and beach seine fisheries in the SWIO.

Country	Gear type	Bycatch estimate
Comoros	Gillnet	Green turtles and hawksbills have mortality as high as 63% (green turtles between 76-89%) 31% of fishers on Grande Comore reported at least one capture and 8.7% on Moheli reported at least one turtle capture[1]. Gillnet fishers that do capture turtles catch on average 30 per year[2]
Kenya	Gillnet	Monofilament: 0.286 indiv.boat^{-1}.y^{-1} , Multifilament: 1.37 indiv.boat^{-1}.y^{-1} Species composition: 65.2 % green turtles, 13 % loggerheads, 13 % hawksbills, 4.3 % leatherbacks, 4.3 % olive ridley, with 31 % mortality[3]
	Bottom-set gillnet	2.51 indiv.boat^{-1}.y^{-1} (species composition as for gillnet), 31 % mortality[3]
	Beach seine	1.33 inidv.boat^{-1}.y^{-1}, 31 % mortality Species composition:53 % green turtles, 21 % hawksbills, 18 % loggerheads, 8 % olive ridleys[3]
Madagascar	Artisanal fishery	11 000 to 15 000 annual mortalities[4] Capture rate across several studies throughout Madagascar of 8.37 indiv.km^{-1}.y^{-1} [5] 73 % capture by gillnets 17.7% by spear or harpoon rest (9.1%) is mixed methods and 0.4% no gear type Species composition: 93.6 % green turtles, 3.4 % hawksbill turtles, 1.6 % loggerheads, 1 % unidentified, 0.4 % olive ridleys[5]
Mauritius	Bottom-set gillnet	15.5 indiv.y^{-1} (hawksbills 75 % and green turtles 25%)[3]
	Beach seine	283 indiv.y^{-1} (47 % green turtles and 53% hawksbills)[3]
Mayotte	Artisanal fishery	111-256 turtle mortalities (18 % of mortalities for net fisheries; 33% of turtles caught in nets died)[6]
Mozambique	Artisanal fishery	240-420 indiv.y^{-1} caught, 75 % are green turtles[7]
	Drift gillnet	0.33 indiv.boat^{-1}.y^{-1} (45 % loggerheads, 20 % green turtles, 20 % hawksbills, 15 % leatherbacks), 8 % mortality[3]
	Bottom set gillnets	0.743 indiv.boat^{-1}.y^{-1} (45 % loggerheads, 20 % green turtles, 20 % hawksbills, 15 % leatherbacks), 8 % mortality[3]
	Beach seine	1.56 indiv.boat^{-1}.y^{-1} (38 % loggerheads, 21 % leatherbacks, 20 % hawksbills, 14 % green turtles), 8 % mortality[3]
South Africa	Gillnet	Loggerheads: 40.90 indiv.year^{-1} (53.3% mortality) Leatherbacks: 5.36 indiv.year^{-1} (63.4 % mortality) Green turtles: 11.9 indiv.year^{-1} (67.2 % mortality) Hawksbills: 1.93 indiv.year^{-1} (72.5 % mortality) Olive ridleys: 0.61 indiv.year^{-1} (65.6 % mortality)[8]

Country	Gear type	Bycatch estimate
Tanzania	Bottom-set gillnet	1.275 indiv.boat^{-1}.y^{-1} (54 % green turtles, 32.6 % loggerheads, 8 % hawksbills, 5 % olive ridleys, 0.4 % leatherbacks), 47 % mortality[3]
	Drift gillnet	Monofilament: 0.143 indiv.boat^{-1}.y^{-1} [3] Multifilament: 0.949 indiv.boat^{-1}.y^{-1} [3] (54 % green turtles, 32.6 % loggerheads, 8 % hawksbills, 5 % olive ridleys, 0.4 % leatherbacks), 47 % mortality[3]
	Beach seine	0.75 indiv.boat^{-1}.y^{-1} (98 % loggerheads, 2 % green turtles)[3]

References

1. Poonian *et al.* (2008)
2. Moore *et al.* (2010)
3. Kiszka (2012a)
4. Rakotonirina and Cooke (1994)
5. Humber *et al.* (2011)
6. Pusineri and Quillard (2008)
7. Louro *et al.* (2006)
8. Brazier *et al.* (2012)

Table 4.9 Summary of annual bycatch of sea turtles in gillnet and beach seine fisheries in the SWIO, giving annual bycatch rates (B, indiv.y^{-1}) and annual mortality rates (M, indiv.y^{-1}) per fishery, per species and country.

Country	**Green turtles**		**Hawksbills**		**Leatherbacks**		**Loggerheads**		**Olive ridley**		**Unidentified**		**Total**	
	B	**M**	**B**	**M**	**B**	**M**	**B**	**M**	**B**	**M**	**B**	**M**	**B**	**M**
Gillnet fishery														
South Africa	12	8	2	1	5	3	41	22	1				61	35
Mozambique	1469	118	1469	118			3305	264	1102	88			7343	587
Tanzania	1594	749	236	111	12	6	963	452	148	69			2953	1388
Kenya	1040	322	207	64	69	21	207	64	69	21	3	1	1594	494
Madagascar	24892	24892	356	356			573	573	48	48	154	154	26022	26022
Comoros	1877	1182	239	150			159	100					2275	1433
Mauritius	4	1	12	4									16	5
Total	**30887**	**27272**	**2520**	**804**	**86**	**30**	**5248**	**1477**	**1366**	**227**	**157**	**155**	**40264**	**29964**
Beach seine fishery														
Mozambique	521	42	744	60			1413	113	781	62	260	21	3719	298
Tanzania	0	0	29	14			1425	670					1454	684
Kenya	184	57	73	23			62	19	28	9			347	108
Madagascar	3019	3019	202	202					96	96	51	51	3368	3368
Mauritius	150	47	133	41									283	88
Total	**3874**	**3164**	**1181**	**339**	**0**	**0**	**2901**	**802**	**904**	**167**	**311**	**72**	**9171**	**4544**
Beach seine and gillnet fisheries combined														
Mayotte	112	37	28	9	0	0	0	0	0	0	0	0	140	46
Total	**34872**	**30473**	**3729**	**1153**	**86**	**30**	**8149**	**2279**	**2270**	**393**	**468**	**227**	**49575**	**34555**

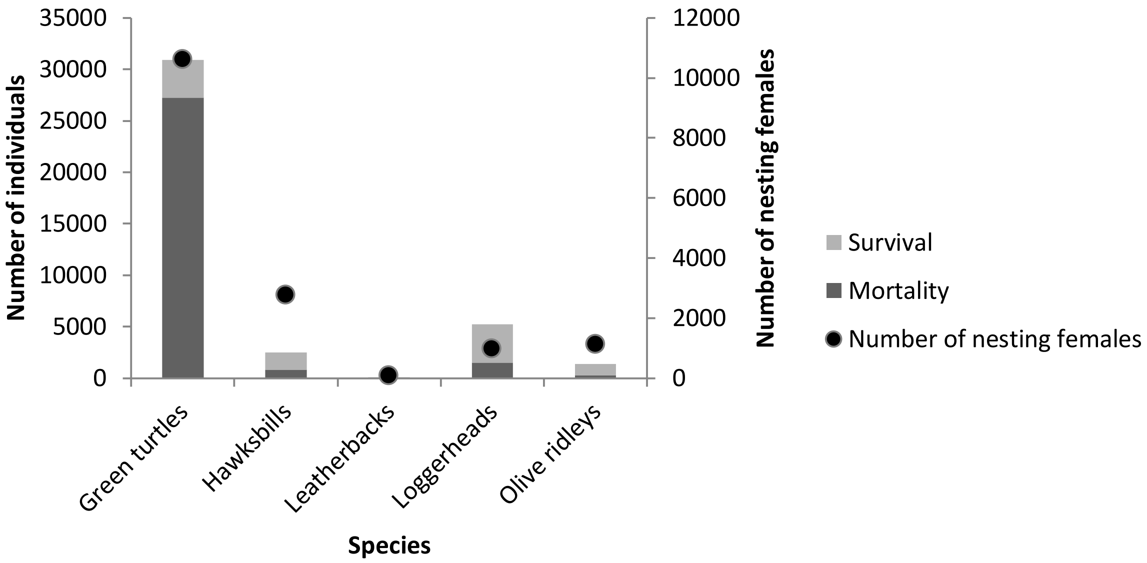


Figure 4.6 Number of sea turtles caught per year in the gillnet fishery in the SWIO, indicating mortality, and survival per species as well as the annual number of nesting females per species per year.

Table 4.10 Mean annual bycatch (indiv.y^{-1}) and mean annual mortality (indiv.y^{-1}) per species in the gillnet fishery. Bycatch and mortality are also expressed as the percentage of the annual number of nesting females in the SWIO (WIO region population size used for olive ridleys).

Species	**No annual nesting females**	**Bycatch**		**Mortality**	
		Total (indiv.y^{-1})	**% of nesting females**	**Total (indiv.y^{-1})**	**% of nesting females**
Green turtles	10599	30887	291.41	27272	257.31
Hawksbills	2762	2520	91.24	804	29.11
Leatherbacks	72	86	119.44	30	41.67
Loggerheads	980	5248	535.51	1477	150.71
Olive ridleys	1114	1366	122.62	227	20.38

Capture rates in the gillnet fishery per country ranged 16 – 26 022 indiv.y^{-1}, with Madagascar (26 022 indiv.y^{-1}) and Mozambique (7 343 indiv.y^{-1}) having the highest capture rates (Table 4.9). The lowest capture rates were in Mauritius (16 indiv.y^{-1}) and South Africa (61 indiv.y^{-1}). The mortality rates in the gillnet fishery varied amongst countries with the lowest mortality rate for Mauritius (~5 indiv.y^{-1}) and the highest for Madagascar (26 022 indiv.y^{-1}; Table 4.9).

Capture rates per species for beach seine fisheries was between 0 and 4 784 indiv.y^{-1}, with the highest capture rates for green turtles (4 784 indiv.y^{-1}) and loggerhead turtles (2 901 indiv.y^{-1}) and the lowest capture rate for leatherback turtles (0 indiv.y^{-1}; Table 4.11 and Fig 4.7). The number of individuals caught per species expressed as a percentage of the number of nesting females varied amongst the species, with the lowest percentage for leatherbacks (0 %) and the highest percentage for loggerheads (296.02 %; Table 4.11). Shapiro-Wilk test indicated that normality is a reasonable assumption for the beach seine bycatch data (W = 0.9295, p = 0.59), thus a Pearson product-moment correlation coefficient was computed to assess the relationship between the bycatch rate and the abundance of species in the region. There is however no significant correlation between the two variables (r = 0.484, n = 5, p = 0.236).

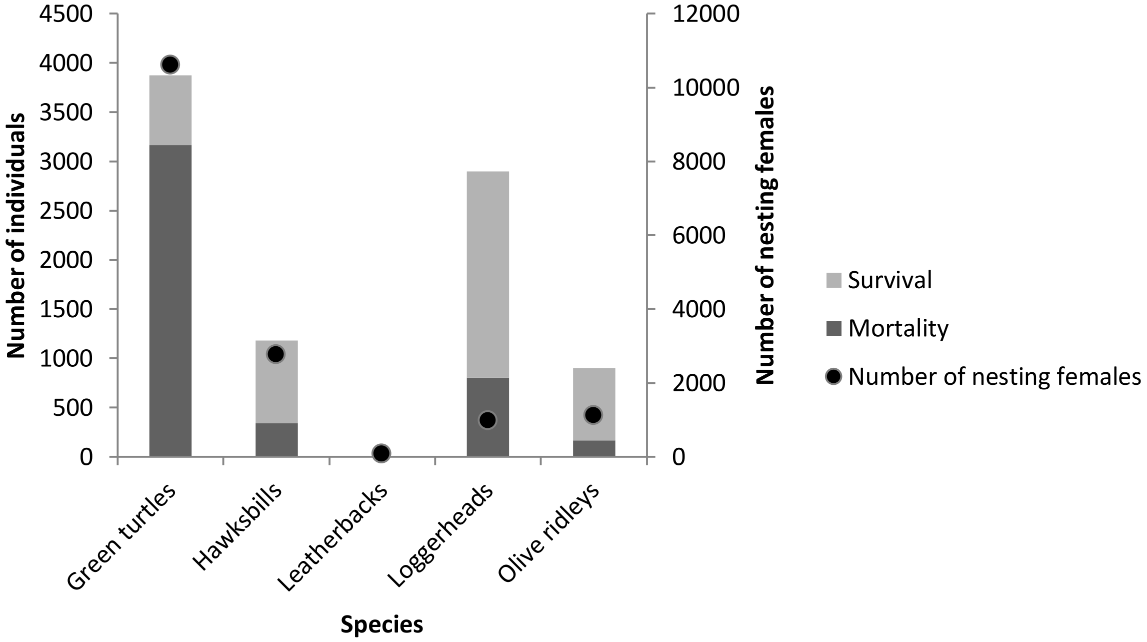


Figure 4.7 Number of sea turtles caught per year in the beach seine fishery in the SWIO, indicating mortality, and survival per species as well as the annual number of nesting females per species per year.

Table 4.11 Mean annual bycatch (indiv.y^{-1}) and mean annual mortality (indiv.y^{-1}) per species in the beach seine fishery. Bycatch and mortality are also expressed as the percentage of the annual number of nesting females in the SWIO.

Species	**No annual nesting females**	**Bycatch**		**Mortality**	
		Total (indiv.y^{-1})	**% of nesting females**	**Total (indiv.$y^{-1)}$**	**% of nesting females**
Green turtles	10599	3874	36.55	3164	29.85
Hawksbills	2762	1181	42.76	339	12.27
Leatherbacks	72	0	0.00	0	0.00
Loggerheads	980	2901	296.02	802	81.84
Olive ridleys	1114	904	81.15	167	14.99

Mortality rates per species for the beach seine fishery ranged from the lowest for leatherbacks (0 indiv.y^{-1}), to the highest for green turtles (3 164 indiv.y^{-1}; Table 4.11). The number of mortalities per species expressed as a percentage of the number of nesting females ranged between the lowest for leatherbacks at 0 %, to the highest for loggerheads at 81.84 % (Table 4.11).

Capture rates per country for the beach seine fishery ranged from 283 to 3 719 indiv.y^{-1}, with Mozambique (3 719 indiv.y^{-1}) and Madagascar (3 368 indiv.y^{-1}) having the highest capture rates and Mauritius (283 indiv.y^{-1}) having the lowest (Table 4.9). The mortality rates also varied amongst countries with the lowest mortality rate for Mauritius (88 indiv.y^{-1}), and the highest mortality rate for Madagascar (3 368 indiv.y^{-1}; Table 4.9).

Spatial overlap between coastal fisheries and sea turtle species

Horizontal overlap

There is limited horizontal overlap between the distribution of sea turtles and coastal fishing operations in the SWIO. The greatest overlap with coastal fisheries was with gillnet fisheries where overlap varied from 1.29 – 1.63 % (Fig. 4.8). The horizontal overlap with prawn trawl fisheries was limited to 0.41 – 0.63 % (Fig. 4.8), with the greatest overlap with leatherbacks (0.63 %) and loggerhead turtles (0.50 %) and the least overlap with hawksbills (0.41 %). The species showing the highest overlap with the gillnet fishery were leatherbacks (1.63 %) and loggerheads (1.55 %), whilst the species showing the least overlap with gillnet fisheries was hawksbills (1.29 %). Beach seine operations and sea turtles overlapped only 0.26 – 0.4 % (Fig. 4.8), also with the greatest overlap with leatherback (0.40 %) and loggerhead turtles (0.32 %) and the least overlap with hawksbill turtles (0.26 %, Fig. 4.8). A Shapiro-Wilk test indicated that normality is not a reasonable assumption for the gillnet data (W = 6861, p = 0.006806). A spearman rank correlation was thus run and a significant moderate negative correlation exists between the two variables (Spearman's rho= - 0.5, D_f = 3, P=0.45, Fig. 4.9). A Shapiro-Wilk test indicated that normality is a reasonable assumption for the beach seine bycatch data (W = 0.9295, p = 0.5932), thus a Pearson product-moment correlation coefficient was computed to assess the relationship between the bycatch rate and horizontal overlap. There is no significant correlation between the two variables (r = 0.493, n = 5, p = 0.143, Fig. 4.10).

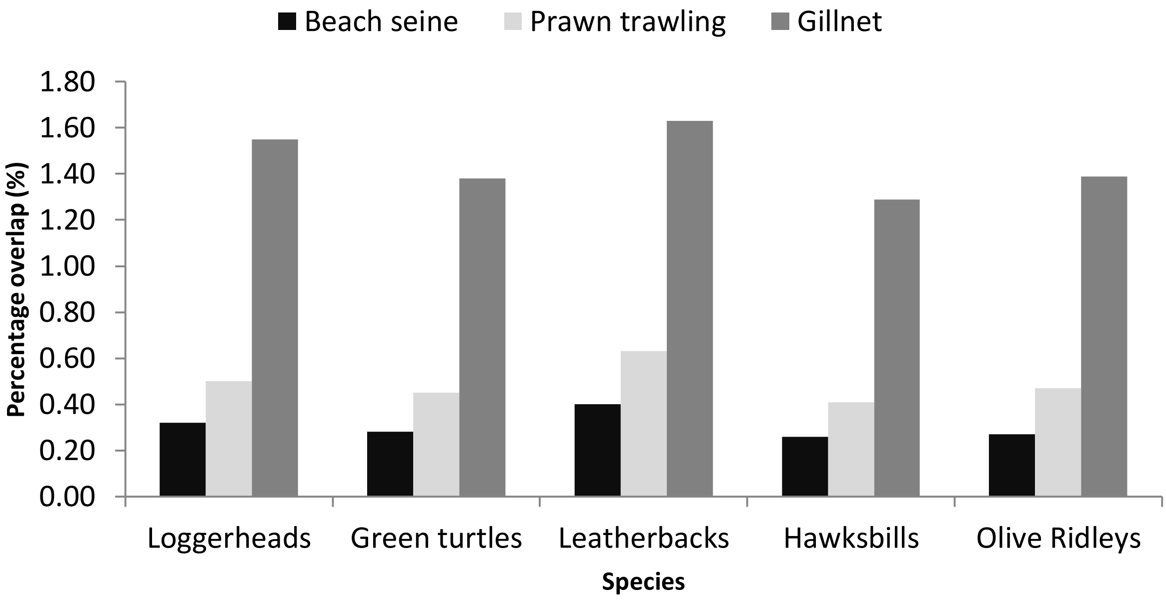


Figure 4.8 Figure showing percentage horizontal overlap between beach seine, prawn trawl and gillnet fisheries and distribution of sea turtle species in the SWIO.

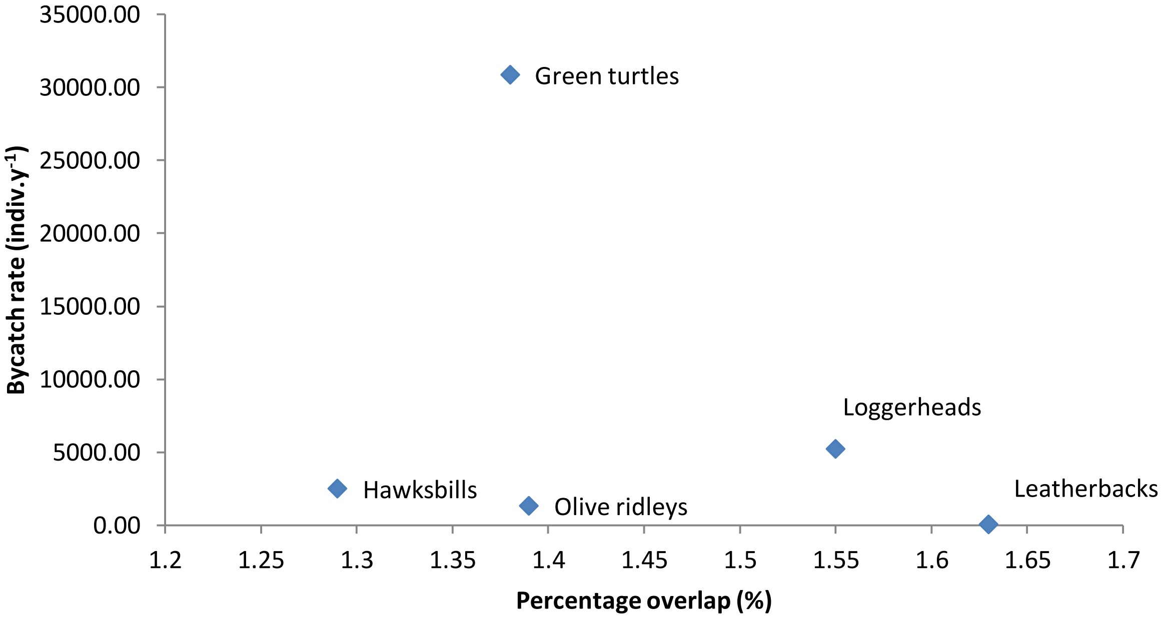


Figure 4.9 Bycatch rates (indiv.y^{-1}) per species expressed as a function of percentage horizontal overlap between sea turtle distributions and gillnet fishery in the SWIO.

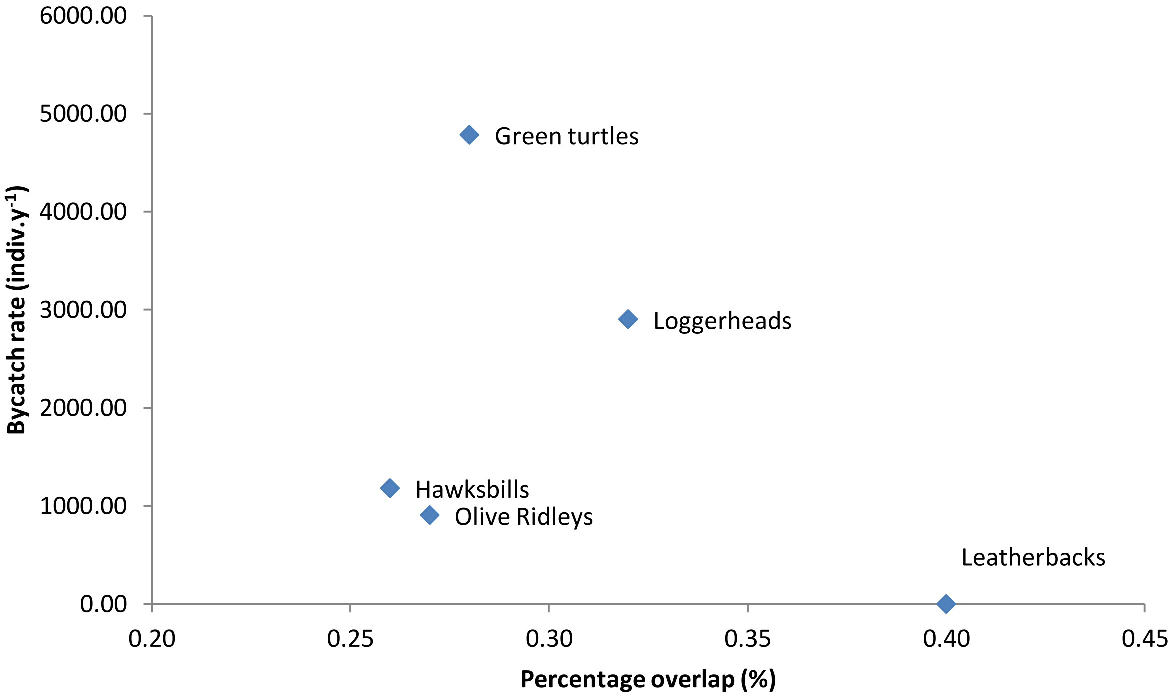


Figure 4.10 Bycatch rates (indiv.y^{-1}) per species expressed as a function of percentage horizontal overlap between sea turtle distributions and beach seine fishery in the SWIO.

Vertical overlap

Maximum diving depth

Sea turtles can access the entire water column (0 – 70 m) in shallow trawling grounds (with a maximum depth of 70 m and typical operational depth of 10 - 70 m), thus for all species there is 86 % overlap between their maximum diving depth and the operational depth of the fishery . For the beach seine fishery with a maximum depth of 30 m there is 100 % overlap between the maximum diving depth and the operational depth of the fishery. Gillnets are set in depths up to 200 m, so all species except hawksbill and green turtles have maximum-recorded diving depths that exceed the depth of the fishery. There is thus complete overlap (100 %) between the gillnet fishery and the maximum depth that sea turtles can dive to in the fishing areas. There is thus no relationship between the overlap and the magnitude of bycatch.

Average diving depths

For the prawn trawl fishery there is substantial variation in overlap between the mean diving depth of sea turtle species and the operational depth of the fishery, with the greatest overlap for olive ridleys (74.9 %) and the least overlap for hawksbills (0 %; Fig. 4.11). For both gillnet and beach seine fisheries there is complete overlap (100 %) between the mean diving depth of all sea turtles (per species) and the operational depth of the fisheries, because the mean diving depth of sea turtles is less than the maximum depth of the fishery.

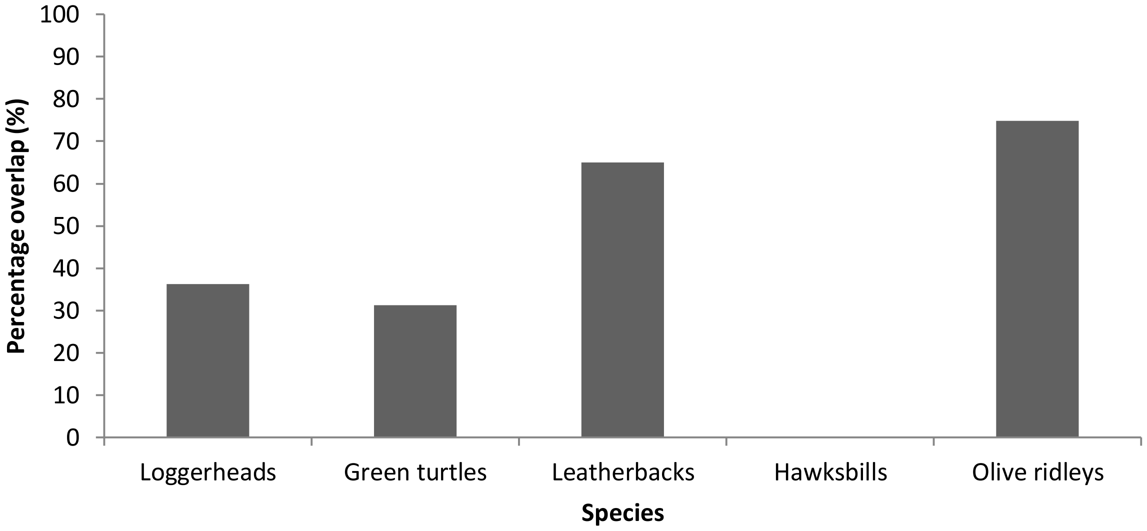


Figure 4.11 Vertical overlap between the mean diving depths of sea turtles and the operational depth of prawn trawl fisheries.

Discussion

The aim of this chapter was to quantify the bycatch of sea turtles in coastal prawn trawl, gillnet, and beach seine fisheries and to determine whether the differences in magnitude of bycatch per species in the region was related to the spatial overlap with the fisheries. The results indicate that gillnet fishery has a higher impact as it has a higher bycatch (40 264 indiv.y^{-1}) and mortality rate (29 965 indiv.y^{-1}) than the prawn trawl (bycatch of 1089 – 2795 indiv.y^{-1}) and beach seine (bycatch of 9 171 indiv.y^{-1}, mortality of 4 544indiv.y^{-1}) fisheries. The beach seine fishery in turn has three times the bycatch of the prawn trawl fishery. So the gillnet fishery captures approximately 77 % of the 52 370 sea turtles captured annually, the beach seine operations catch 18 %, and prawn trawling catch 5 %. Within the SWIO region the higher magnitude of bycatch in the gillnet fishery in the region can be ascribed to the higher overlap with sea turtles species than for the other fisheries, as well as the unselective nature of the fishing gear (Hamley, 1975). In the Caribbean and Eastern Indian Ocean gillnet fisheries also had higher capture rates than the prawn trawl fisheries (Wallace *et al.*, 2010b). In the Gulf of Mexico (Finkbeiner *et al.*, 2011), Mediterranean (Casale, 2011), and USA (Moore *et al.*, 2009), trawl fisheries were responsible for higher capture rates of sea turtles than gillnet fisheries. These differences can be ascribed to the different efforts that are exerted in fisheries across the different regions, as well as the densities of sea turtles within the fishing areas. Very few data exist for captures of sea turtles in beach seine fisheries outside of the SWIO, making comparison of the impacts from the fishery with other regions impossible (Tietza *et al.*, 2011).

The results suggest that the spatial extent of the three coastal fisheries in the SWIO differ as they use different habitats, but with some partial overlap amongst them. Gillnet fishing areas in the SWIO cover approximately 300 000 km^2 of the region followed by prawn trawl fisheries that cover approximately 65 000 km^2 and beach seine fisheries covering 60 000 km^2. Even though these fisheries have a limited distribution along the coast, all three overlap with sea turtle distributions in the region. Moreover, the spatial overlap occurs in areas with high densities of sea turtles along the coast. It is therefore not the extent but rather the location that is responsible for these fisheries to have substantial impacts on sea turtle populations in the region.

The most commonly captured species in this study in all three fisheries were green turtles and loggerheads, although prawn trawlers also frequently caught hawksbills. In Peru (Alfaro Shigueto *et al.*, 2011, Alfaro Shigueto *et al.*, 2010) and Mexico (Mancini *et al.*, 2011), green turtles were also the most commonly caught species in the gillnet fishery, but the US mid-Atlantic fleet caught mostly loggerheads (Murray, 2009), and leatherbacks are caught in the West Indies in Trinidad and Tobago (Lum, 2006). The Northern Australian prawn trawl fishery, has the highest interactions with olive ridleys and green turtles (Brewer *et al.*, 2006), and in Queensland, Australia, trawl fisheries caught green turtles (Robins, 1995). The differences in capture rates amongst species can be ascribed to difference in abundances of these sea turtles within the fishing areas across the regions. There are very large rookeries of each of these species in close proximity to the fishing areas.

The species expected to have the highest rates of bycatch in the prawn trawl fishery is green turtles, loggerheads, and hawksbills. Although it was not possible to determine species-specific bycatch estimates, the available data indicate that green turtles are most frequently reported in the prawn trawl fisheries (Joynson-Hicks & Ngatunga, 2009, Van der Elst, 2012, Wamukoya & Mbendo, 1995), followed by loggerheads and hawksbills (Joynson-Hicks & Ngatunga, 2009, Van der Elst, 2012). Leatherbacks are reported as bycatch however as a fraction of the other sea turtle species such as loggerheads (Nel *et al.*, 2013a). The prominence of green turtles, hawksbills, and loggerheads in trawl captures compared to leatherbacks can be attributed to two factors. Firstly, these species are more abundant in the region than leatherbacks and secondly these species tend to remain in coastal waters as adults and sub adults whereas leatherbacks prefer both offshore and coastal waters for feeding. Leatherbacks however do use coastal waters as inter-nesting habitat. Leatherback interactions were recorded in prawn trawl fisheries along the nesting areas (in both the South African (De Wet, 2012, Nel *et al.*, 2013a) and Mozambican (Gove *et al.*, 2001) prawn trawl fisheries) and few, if any, interactions are recorded in areas away from nesting beaches (Joynson-Hicks & Ngatunga, 2009). It is expected that leatherback turtles that are caught in the prawn trawl fishing operations in the SWIO are mainly reproductively mature adults that interact with these fisheries when they aggregate at nesting beaches along the South African and Mozambican coast during the nesting season, although that seems to be in very low numbers. However, there are still some instances where leatherback turtles are caught in the fishery away from nesting beaches, however this is expected to be in very low numbers.

The gillnet fishery is expected to have the greatest impact on the green turtle and leatherback populations in the SWIO. In the region a large number of green turtles are capture in the gillnet fishery (30 887 indiv.y^{-1}) with an associated mortality of 27 272 indiv.y^{-1}. The number of bycaught green turtles in the region amount to almost three times the number of nesting females in the region with the mortality of green turtles being approximately two and a half times the number annual nesting females. Thus, both the bycatch rate and mortality rate for this species is high compared to the population size. Compared to green turtles the number of leatherbacks that are captured in the region (86 indiv.y^{-1}) is small, however this low capture rate is still very high compared to the population size (more than 110 % of the number of nesting females). The number of mortalities for leatherbacks are also low (30 indiv.y^{-1}) compared to green turtles; however it is high in relation to the leatherback population size (> 41 % of the number of annual nesters). The capture (5248 indiv.y^{-1}) and mortality (1477 indiv.y^{-1}) rates of loggerheads in the fishery is high compared to the number of nesting females (535.51 % and 150.71 % respectively). Green turtles, loggerheads, and leatherbacks are receiving protection at selected nesting beaches in the region (Bourjea *et al.*, 2007a, Lauret-Stepler *et al.*, 2007, Nel *et al.*, 2013a). Loggerheads have shown an increase in population numbers over several decades, whereas the leatherback population has shown a stable population trend recently (Nel *et al.*, 2013a), and green turtles have also shown a recent increase in population numbers (Lauret-Stepler *et al.*, 2007). The increase in the green turtle population in the region is, however, not as high as expected (Bourjea, 2012). It is thus expected that the gillnet fishery is partly responsible for the slowing of the recovery rate of the leatherback and green turtle populations. Despite the high capture rates in this fishery the loggerhead population is still increasing indicating that there is are other factors at play in the high bycatch rate.

The beach seine fishery has the greatest impact on the green turtle population in the region due to the high magnitude of bycatch (3 874 indiv.y^{-1}) and mortality (3 164 indiv.y^{-1}) of this species in the fishery. However, the catches do not amount to a large proportion of the population size (36.55 % and 29.85 % of the number of annual nesters respectively). The bycatch (2 901 indiv.y^{-1}) and mortality (802 indiv.y^{-1}) for loggerheads in the beach seine fishery in the SWIO amount to 296.02 and 81.84 % of the annual number of nesting females. Both the green and loggerhead populations have shown recent increases in population numbers (Bourjea *et al.*, 2007a, Lauret-Stepler *et al.*, 2007, Mortimer *et al.*, 2011b, Nel *et al.*, 2013a), however the increase in the green turtle population is not as high as expected (Bourjea, 2012), indicating that the beach seine fishery might be cumulatively adding to the effect of the gillnet fishery in slowing the population growth rate of this species in the region. Despite the high capture rates in the fishery the loggerhead population has shown significant increases in numbers over at least 5 decades (Nel *et al.*, 2013a). The high capture rates of loggerheads in gillnet and beach seine fishery would be expected to have an impact on the population considering that the capture rates exceed the annual number of nesting females. There is overlap between the loggerhead populations from the SWIO RMU with the Arabian Gulf (NWIO) RMU at feeding grounds within the Mozambique Channel (Dalleau *et al.*, 2014). The highest magnitude of loggerhead captures is seen in Mozambique and Tanzania in both these fisheries, where it is expected that there is higher overlap between these two RMUs. The size of the Arabian Gulf RMU at ~30 000 nesting females per annum (Nichols, 2007) is thus several times larger than that of the SWIO RMU. This overlap in distribution between RMUs of different size classes explains why high capture rates of loggerhead turtles in the region have not decimated the population in the SWIO. It also confirms that the proportion of the captures is a function of population size, which in this case can presumably be assigned to individuals from the Arabian Gulf RMU. It also suggests that the boundaries of the RMUs are not as discrete as is suggested by Wallace et al (2010).

The capture rates amongst species in the gillnet fishery show a positive correlation with the abundance of species in the region, with the bycatch rate increasing as the abundance of a species increases. However, the same is not true for the beach seine fishery where there is no significant relationship between the bycatch rate and abundance. For both the beach seine fishery there is no significant relationship between the bycatch rate and horizontal and vertical overlap. The gillnet fishery however shows a significant moderate correlation between the bycatch rate and horizontal overlap; however no significant relationship exists for vertical overlap. Other factors such as the sea turtle abundances at nearby rookeries may play an important role in the magnitude of the bycatch in the region. It is expected that encounters with coastal fishing gear will be chance encounters and that the captures will be a reflection of the abundance of sea turtles in the coastal habitats, because there is no indication that sea turtles may be attracted to either of the gear types used (Gilman *et al.*, 2010b).

Despite the fact that this study is the most comprehensive study (of existing data) on the impacts of coastal fisheries on sea turtle populations in the SWIO, there are several limitations to the study. The data limitations that exist concerning fishing pressure and capture rates of sea turtles in the region forced simplistic analysis. The estimates are thus conservative estimates of the impact fisheries in the region but may also totally overestimate catches when it is raised to the fishery level. The methods did not take into account the multitude of factors that can have an effect on sea turtle capture rates (such as fine scale differences in sea turtle densities and how these overlap with differing fishing areas within a country as this information was largely unavailable for the species and the fisheries). The differences in capture rates amongst gear configuration and setting methods were also largely ignored, as there are no data available for this type of analysis. Some of the estimates for the prawn trawl fishery may be over-estimating current bycatch rates (e.g. Kenya and Mozambique) due to the subsequent implementation of TEDs. However, this cannot be corrected for because there are no new data available for these countries. The estimates of mortality in the fisheries also do not include post-release mortality, as there is no information available regarding this. The post-release mortality is also dependent upon the injury status of the individual sea turtle upon release and without definitive information available on the status upon release; such an estimate cannot be made.

Despite the high numbers of sea turtles caught in these fisheries, it is possible to mitigate the bycatch (Appendix C). However, bycatch mitigation in artisanal fisheries may present some difficulty due to the value of sea turtles caught for the fishermen either as a source of nutrition, medicinal purposes or as a source of income. There is thus higher mortality rates in artisanal fisheries compared to those in industrial fisheries (see Chapter 3). The lack of management of the artisanal fisheries in the SWIO (Kimani *et al.*, 2009) indicates that there will be significant difficulty to enforce management controls aimed at reducing bycatch in these fisheries. In order to address the bycatch in artisanal fisheries is will be necessary to understand the socio-economic and cultural environments of the communities (especially the value placed on turtles as a food source and source of income) that partake in fishing operations in order to identify possible means of reducing bycatch. Conservation based community development programs such as the South African turtle monitoring program have substantial value in aiding conservation whilst producing employment for the local community.

Chapter 5

Productivity-Susceptibility Analysis (PSA) for sea turtles in selected fisheries in the SWIO region

Abstract

Productivity-Susceptibility Analyses are frequently used in fisheries to evaluate the relative vulnerability of bycatch species to fisheries in data-poor situations. These analyses can provide a guide for prioritizing species or populations for protection, and highlight specific threatening activities. In this Chapter, existing PSA methods, commonly used to assess vulnerability of bycatch species, were modified to suit the life history of sea turtles and fisheries in the SWIO. Gillnet, beach seine and longline fisheries were identified as posing the greatest threat to sea turtle populations in the SWIO. The main results indicate that gillnet fisheries pose the greatest threat to sea turtle populations in the SWIO specifically to leatherback and green turtle populations, whereas longline fisheries pose a substantial threat to loggerhead and leatherback populations. A cumulative impact assessment indicates that the species that are most vulnerable to the combined threat posed by the fisheries assessed are leatherback and loggerhead populations. The loggerhead and green turtle populations in the SWIO are however recovering, whereas the leatherback population is currently stable (but not recovering despite conservation). The bycatch of leatherback turtles is thus a concern, as there is possibly a substantial source of mortality that has not yet been identified.

Introduction

The SWIO region hosts five species of sea turtles, all of them are listed as endangered by the IUCN (Table 1.1, IUCN, 2014). Several of these species are well protected at selected nesting beaches in the SWIO (Mortimer *et al.*, 2011b, Nel *et al.*, 2013a). However, most of these populations are not recovering as expected, regardless of their protection, particularly the leatherback population in South Africa (Nel *et al.*, 2013a). One of these factors in their marine environment that might be hampering the recovery of sea turtle populations, is the fishing pressure faced by these populations. In the SWIO there are a host of fisheries that overlap with sea turtle populations that have the potential to impact severely on these populations, including longline (Petersen *et al.*, 2009), purse seine (Clermont *et al.*, 2012), prawn trawl (Gove *et al.*, 2001), gillnet (Humber *et al.*, 2011, Poonian *et al.*, 2008) and beach seine (Humber *et al.*, 2011, Kiszka, 2012b) fisheries. It is essential to identify the fishery or fisheries that have the greatest impact on sea turtle populations as well as the species that are most impacted by these fisheries in order to effectively conserve these species across the marine environment.

Using quantitative methods to assess the impact of fisheries requires substantial knowledge of the aforementioned factors (see Chapter 2) especially survivorship data (Heppell, 1998). Thus, despite our knowledge that these fisheries affect sea turtle populations, quantifying the impact is difficult because there are substantial gaps in the data on fishing effort, sea turtle bycatch in the fisheries and sea turtle demographics. Gaps in the data for the artisanal fisheries are more notable than for industrial fisheries (as seen in Chapter 3 and 4). However, this difficulty can be overcome by using semi-quantitative methods such as Productivity-Susceptibility Analysis (PSA). The PSA methodology was originally developed to determine the differences between the sustainability of bycatch in the Australian prawn trawl fishery (Milton, 2001, Stobutzki *et al.*, 2001), and has since been used in several fisheries to assess the vulnerability of bycatch species to fishing practices (Braccini *et al.*, 2006, Griffiths *et al.*, 2006, Patrick *et al.*, 2010, Stobutzki *et al.*, 2001, Zhou & Griffiths, 2008). PSAs are widely applicable where there are gaps in information (Zhou *et al.*, 2011) and are recommended by several organisations and working groups as a reasonable approach to determine relative risks (Hobday *et al.*, 2007, Rosenberg *et al.*, 2007, Smith *et al.*, 2007).

In a PSA, the productivity of a species (proxy for the intrinsic rate of increase) is assessed in relation to the susceptibility (proxy for the likelihood of a species to be caught) in a fishery. The productivity assessment in a PSA uses demographic characteristics such as longevity, fecundity, recruitment, and natural mortality as a measure of the intrinsic growth rate (r) of a population. This provides an indication of the population's relative resistance to fishing mortality and its ability to recover from depletion. Susceptibility, on the other hand, uses fishery-related factors to determine the probability of a species to be caught, such as the spatial and temporal overlap between fisheries and species, the probability of their interaction, and the probability of capture, injury, and mortality because of interaction with the fishery. These productivity and susceptibility scores are then used to calculate a vulnerability score that indicates the likelihood that a species may be impacted by a particular fishery (Ormseth & Spencer, 2011).

The aim of this Chapter is to assess the relative vulnerabilities of sea turtle populations to fisheries in the SWIO. In order to achieve this aim there are several main objectives. Firstly, to develop a sound ERA methodology in the form of a PSA that is specifically applicable to sea turtles. Secondly, to perform a sensitivity analysis on the process to determine how sensitive it is to changes. Thirdly to assess the relative productivity of sea turtle populations in the SWIO. Fourthly, to assess the relative susceptibility of sea turtle populations to fisheries in the SWIO. Fifthly, to assess the relative vulnerability of sea turtle populations to these fisheries. Lastly, to assess the cumulative effect that these fisheries have on sea turtle populations.

Methods

The PSA method used here is based on a procedure developed by Milton (2001), and modified by (Arrizabalaga *et al.*, 2011) and Ormseth and Spencer (2011). It was first applied to sea turtles in the Indian Ocean by Nel *et al.* (2013b). In this method the productivity of a population is assessed based on the life history characteristics to rank the population according to their Productivity score (P). The Susceptibility score (S) reflects the likelihood of capture and mortality in a particular fishery. Each of the attributes used in the calculation of the scores are assigned a value according to ranks ranging from low (1) to high (3). Representing these on a graph with productivity on the x-axis and susceptibility on the y-axis, makes it possible to calculate a vulnerability score for each population in relation to a fishery. The vulnerability score is then used to rank to populations and fisheries relative to each other. The PSA procedure used in this Dissertation is explained in detail below.

Productivity analysis

The productivity attributes used in this assessment includes direct measures of population viability (e.g. longevity, fecundity and recruitment), and other factors (e.g. population trends) (Table 5.1, Appendix D). The recovery potential of a species depends highly on the number of hatchlings produced per year. The number of hatchlings produced per population in turn is dependent on the size of the nesting population, fecundity factors, and nest success. Another important factor that influences the recovery potential is the time it takes an individual to reach sexual maturity. The longer it takes to reach sexual maturity the higher the chances of an individual dying prior to breeding. Within the reproductive lifespan of a female sea turtle, the expectation is that she will return to a nest several times, generally two to three years apart. An individual with a longer reproductive lifespan is thus more productive as there is a greater possibility for more nesting events than in individuals/species with a shorter reproductive lifespan. Frequency of breeding should also be considered. Two individuals/species with similar reproductive lifespans but with different nesting frequencies (or remigration periods in the case of sea turtles) will have different reproductive outputs (keeping all other factors constant). The generation length of a species is an indication of the turnover rates of individuals in the population thus the longer the generation length the lower the productivity of the species (IUCN, 2001).

Data for each of the productivity attributes were obtained from published literature, unless otherwise stated. Information for the productivity criteria was available for most of the populations, except olive ridleys for which very little information exists (See Chapter 2). Data inferred from adjacent rookeries were applied with caution as it may be population specific and affected by distance to the foraging ground, which may affect the nesting biology such as remigration intervals (Limpus, 2008a, Limpus, 2008b). Species-specific information such as the number of eggs laid per female per clutch, which is generally consistent (Limpus, 2007), was applied without caution because values were assigned a broad rating and actual numbers were less important. Each of the attributes was scored as high (3), medium (2) or low (1) productivity. When no information was available for a specific attribute and drawing inferences was not possible, the attribute was assigned low (1) productivity score using the precautionary principle. Weightings were assigned to each attribute based on the perceived importance of the criteria in determining population viability, and confidence in the data. The population size is the most important attribute for which the most reliable data were available and was thus assigned the heaviest weighting (3). Trend data was weighted the next highest (2) due to the reliability of the data. In addition, if a population was already in decline any additional mortality can have a substantial impact on the population. All other criteria were given equal weightings (1). The overall productivity score was obtained as a weighted average of the scores of individual criteria as follows:

$$\mathbf{P} = \frac{\sum_{\mathbf{i=1}}^{\mathbf{n}} \mathbf{w_i s_i}}{\sum_{\mathbf{i=1}}^{\mathbf{n}} \mathbf{w_i}} \qquad \textbf{Equation 5.2}$$

Where n is the number of criteria, w_i is the weighting of the i^{th} criterion and s_i is the score of the i^{th} criterion.

The weighted standard deviation (SD_w^2) was calculated as follows:

$$\mathbf{SD_w^2} = \sqrt{\frac{\mathbf{N'\sum_{i=1}^{N} w_i(s_i - P)^2}}{\mathbf{(N'-1)\sum_{i=1}^{N} w_i}}} \qquad \textbf{Equation 5.3}$$

Where N' is the number of non-zero weights.

Overall productivity was considered high when the score was above 2.32, medium when the score ranged from 1.66 – 2.32, and low when the score was below 1.66.

Table 5.1 Productivity attributes used in the analysis, including the scoring and weighting information (See Appendix D for details on each of the attributes).

Productivity criteria	**Score**	**Weight**
1. Long term population trend	1 Uncertain or decline 2 Stable 3 Increase	2
2. Recent population trend	1 Uncertain or decline 2 Stable 3 Increase	2
3. RMU size (Population size)	1 Very small 1.5 Small 2 Medium 2.5 Large 3 Very large	3
4. Genetic diversity	1 Low (1 genetic stock) 2 Medium (2 genetic stocks) 3 High (> 2 genetic stocks)	1
5. Age at maturity	1 Low (>25 years) 2 Medium (15 – 25 Years) 3 High (<15 years)	1
6. Reproductive lifespan	1 Low (>20 years) 2 Medium (18-20 years) 3 High (<18 years)	1
7. Generation length	1 Low (>43 years) 2 Medium (26-43 years) 3 High (<26 years)	1
8. Remigration Interval	1 Low (>4 years) 2 Medium (4-2.6) 3 High (<2.6)	1
9. Number of clutches per female per season	1 Low (<4 nests) 2 Medium (4-6) 3 High (>6)	1
10. Number of eggs per female per clutch	1 Low (<90 eggs) 2 Medium (90-120) 3 High (>120)	1
11. Natural mortality: Nest Success	1 Low (<50%) 2 Medium (50 – 75%) 3 High (>75%)	1
12. Natural mortality: Emergence Success	1 Low (<50%) 2 Medium (50 – 75%) 3 High (>75%)	1

Susceptibility analysis

The susceptibility analysis used fishery-related factors to determine the likelihood of capture and mortality in a fishery (Table 5.2 Appendix D). Data on susceptibility attributes were calculated in Chapter 3 and 4. Susceptibility scores were assigned to each of the attributes, scoring high susceptibility as 3, medium susceptibility as 2 and low susceptibility as 1. The mortality estimate was assigned a higher weighting (3), while all other attributes were assigned equal weightings (1). The susceptibility score (S) was calculated as a weighted average of the individual scores of susceptibility criteria using the following equation:

$$\mathbf{S} = \frac{\sum_{i=1}^{n} w_i s_i}{\sum_{i=1}^{n} w_i} \qquad \textbf{Equation 5.4}$$

Where n is the number of criteria, w_i is the weighting of the i^{th} criterion and s_i is the score of the i^{th} criterion.

The weighted standard deviation (SD_w^2) was calculated using the following equation:

$$\mathbf{SD_w^2} = \sqrt{\frac{N' \sum_{i=1}^{N} w_i (s_i - S)^2}{(N'-1) \sum_{i=1}^{N} w_i}} \qquad \textbf{Equation 5.5}$$

Where N' is the number of non-zero weights.

Overall susceptibility is high when the score is above 2.32, medium when the score is range from 1.66 and 2.32 and low when below 1.66.

Table 5.2 Susceptibility criteria used in the analysis, including scoring and weighting information (See Appendix D for details on each of the attributes).

Susceptibility	Score		Weight
1. Horizontal overlap with fishery	1	Low (<30 % overlap)	1
	2	Medium (30-60 % overlap)	
	3	High (>60 % overlap)	
2. Vertical overlap with fishery	1	Low (<30 % overlap)	1
	2	Medium (30-60 % overlap)	
	3	High (>60 % overlap)	
3. Confidence estimate in distribution data	1	Low (<5)	1
	2	Medium (5-30)	
	3	High (>30)	
4. Geographic concentration	Data deficient - not scored		
5. Bycatch mortality relative to natural mortality	1	Low (30%)	3
	2	Medium (31 – 100%)	
	3	High (>100%)	
6. Post-capture mortality	Data deficient - not scored		
7. Seasonal changes in overlap	Data deficient - not scored		
8. Size class most frequently caught	Data deficient - not scored		

Data quality scores

Due to the ranking of criteria and the sizes of bins that are used for the ranking of criteria, small changes in the biological parameters may not have any change on the score that is assigned to an attribute, unless the original values were close to the cut-off point of the bins. This allows leeway in the use of lower quality data that will still result in robust analysis, especially where the bins are larger. Where bins are smaller, as for genetic diversity in this analysis, a small change in the initial biological data used in the analysis will result in an immediate shift of categories. Where bins are relatively small or compact and small changes in data can result in ranking shifts, these criteria should be given higher weightings only when the confidence in the data is high. Due to the uncertainty of data of some parameters, it is useful to assess the quality of data that are used in an analysis by calculating a data quality score. The data quality score will indicate the confidence in the outcomes of the analysis.

Errors in risk estimation can occur where there is data paucity such as for olive ridley stocks in the SWIO (Astles *et al.*, 2006). Where data are unavailable for a criteria, as a precautionary measure the criteria are scored as high susceptibility or low productivity, in order to avoid assigning low risk to a stock that in fact has a high risk (Astles *et al.*, 2006, Milton, 2001, Stobutzki *et al.*, 2001). The risk scores of data poor stocks are thus often inflated (Hobday *et al.*, 2007). It is thus necessary to consider data paucity and missing data in the larger context. Where a stock has very few criteria with missing or low quality data, the effect on the overall risk score would be small. However, when a stock has very limited information and most criteria are data deficient the effect on the overall risk score can be substantial. It was thus necessary to take the data quality of both the productivity and susceptibility scores into consideration when interpreting the overall vulnerability results. Each of the criteria in the assessment was thus assigned a data quality score between 1 (best data) and 5 (no data) (Table 5.3). Data quality scores were computed for productivity, susceptibility, and vulnerability scores. Overall data quality scores were divided into three groups (low >= 3.5, medium 2.0 to 3.49, and high < 2.0). Data quality scores (Q) were computed using the following equations:

$$\boldsymbol{Q = \frac{\sum_{i=1}^{n} w_i s_i}{\sum_{i=1}^{n} w_i}}$$ **Equation 5.6**

Where n is the number of criteria, w_i is the weighting of the i^{th} criterion and s_i is the score of the i^{th} criterion.

With the weighted standard deviation (SD_w^2) calculated using the following equation:

$$SD_w^2 = \sqrt{\frac{N' \sum_{i=1}^{N} w_i (s_i - Q)^2}{(N'-1) \sum_{i=1}^{N} w_i}} \qquad \textbf{Equation 5.7}$$

Where N' is the number of non-zero weights.

Table 5.3 Scores used for the data quality of productivity and susceptibility criteria.

Score	Description
1	The information is based on data collected for the stock and area of interest
2	Data with limited coverage and corroboration
3	Estimates with high variation and limited confidence, may be based on similar taxa or inferred from other populations
4	Expert opinion or based on general literature review or from outside the region
5	No data

Vulnerability Analysis (PSA Analysis)

Before the PSA evaluation was executed, the sensitivity of the PSA to changes in the number of attributes and changes in attribute scores was tested (see Appendix E for details). The productivity and susceptibility scores obtained from the respective analyses were represented as the reversed productivity score (P) on the x-axis and the susceptibility score (S) on the y-axis. The overall vulnerability score (V) was calculated as the Euclidian distance from the origin of the graph using the following equation:

$$V = \sqrt{(P-3)^2 + (S-1)^2} \qquad \textbf{Equation 5.8}$$

This vulnerability scores provides a quantitative measure of the overall vulnerability (Ormseth & Spencer, 2011). These vulnerability scores are used to rank the species according to vulnerability to each of the fisheries (within fishery comparisons) and to rank the vulnerability according to species and fisheries combined (between fisheries comparisons). Overall vulnerability is assessed as high risk when the score is greater than 1.89, medium when the score ranges from 0.94 to 1.89 and low risk when the score is below 0.94. The maximum vulnerability that a species can have is 2.83.

Assessment of Cumulative Effects

In a multi species, multi fishery situation, the mere analysis of the vulnerability per fishery does not give a clear indication of the overall pressure that fishing with multiple gear types is exerting on a particular species within a region. It is thus necessary to conduct a cumulative effects assessment combining all the factors to determine the highest overall vulnerability from the combination of factors. Cumulative effects were assessed based on the methods used by Micheli *et al.* (2014). The approach followed in this Dissertation takes into account that the cumulative impact of fisheries may be larger than that of the fishery that impacts most severely on a species (Halpern *et al.*, 2008). Micheli *et al.* (2014) developed a method to assess the additive impact of fisheries using aggregated susceptibility (AS) that is calculated as follows:

$$\boldsymbol{AS = 1 + \sqrt{(S_{f1} - 1)^2 + (S_{f2} - 1)^2 + (S_{f3} - 1)^2 + \cdots + (S_{fi} - 1)^2}} \qquad \textbf{Equation 5.9}$$

Where i is the number of fisheries assessed. The overall vulnerability in this case is calculated using the AS score as the susceptibility score in Equation 5.7. The cumulative effects assessment was done for two different scenarios. The first scenario used the susceptibility scores of each of the fisheries. In the second scenario the cumulative effects of all impacts were assessed. These impacts included fisheries (using susceptibility scores excluding other impacts) and the other impacts (Table 5.4). Alternative threats (excluding other fisheries) faced by sea turtle populations in the SWIO were scored using data from Wallace *et al.* (2011b). Wallace *et al.* (2011b) scored the threats to sea turtle RMUs using five threats to sea turtle species; (i) fisheries bycatch (ii) take (consumptive use), (iii) coastal development, (iv) pollution and pathogens and, (v) climate change (Table 5.4). The threat scores were scored as relative impact to each of the RMUs, with all the threat scores increasing from 1 (low) to 3 (high). If insufficient information was available for a criterion, it was scored as data deficient and scored as high threat (3; See Wallace *et al.* (2011a) for more information). The threat scores assigned by Wallace *et al.* (2011a) for four of the criteria were used in the calculation of the alternative threat score. These criteria included take, coastal development, pollution, and climate change.

Table 5.4 Threat scores for non-fishery related threats affecting sea turtles populations in the SWIO (after Wallace *et al.* (2011b).

Species	**Take**	**Coastal Development**	**Pollution**	**Climate Change**	**Average**
Loggerheads	1.5[1,2]	1[3]	1[4,5]	1[3]	1.13
Green turtles	2[1,2,6]	1.5[6,7]	1[4,5,6]	2[8]	1.63
Leatherbacks	1[1,2,9]	1[3]	1[9]	1[3]	1
Hawksbills	2.5[10,11]	2[10,11]	1[10,11]	3[10,11]	2.13
Olive ridleys	3[12,13]	3[12,13]	2[12,13]	DD(3)	2.75

References:
1. Bourjea *et al.* (2008)
2. Petersen *et al.* (2009)
3. Harris (2008)
4. Ramessur (2002)
5. Salm (1983)
6. Francis and Bryceson (2000)
7. Bertrand *et al.* (1986)
8. Bourjea (in prep) in Wallace *et al.* (2011a)
9. Luschi *et al.* (2006)
10. Mortimer (2007)
11. Mortimer and Donnelly (2007)
12. Abreu-Grobois (2006)
13. Plotkin (2007)

Results

Data quality

The overall data quality scores for productivity attributes of all species except olive ridleys were high, with mean scores ranging between 1.00 ± 0.00 (SD) – 3.06 ± 1.97 (SD). The best data were available for loggerheads, leatherback and green turtles (Fig 5.1, Appendix F), but data were particularly limited for olive ridleys. The low data quality for olive ridleys can be attributed to the fact that there are few data available regarding the biology of the species in the region (see Table 5.5 for details). The olive ridley RMU was subsequently eliminated from further analysis due to the low confidence in the productivity score (but flagged as a species of concern that should be prioritized for data collection).

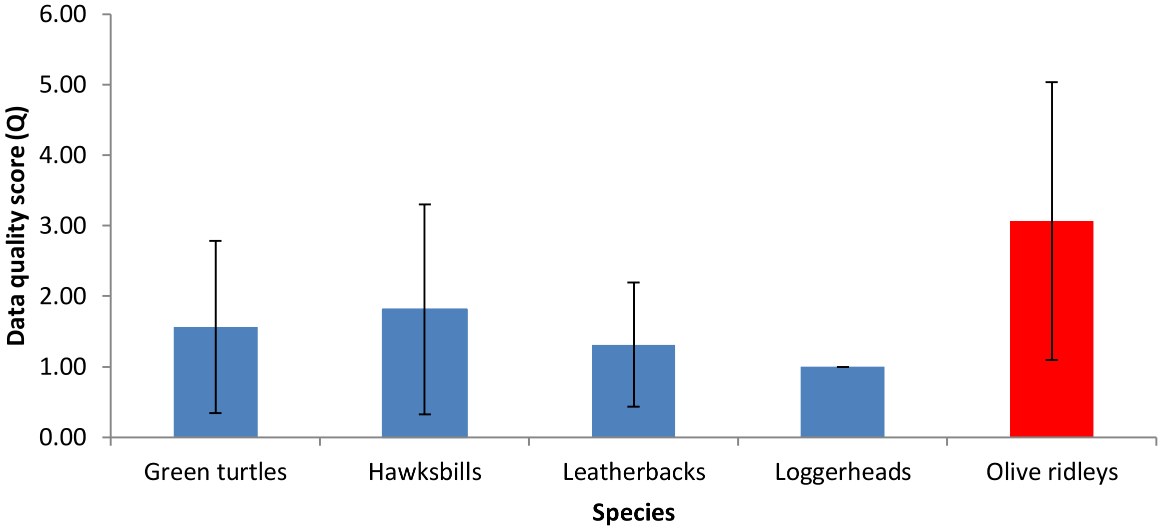


Figure 5.1 Data quality scores for productivity criteria of sea turtle species. Error bars indicate weighted standard deviation. Scores shown in blue are high data quality, and red colored scores are medium quality.

The data quality for the susceptibility analysis of the longline and purse seine fisheries were high for all species, whilst data quality for the other fisheries were all medium quality across all species. The data quality scores ranged between 1.83 ± 0.79 – 2.83 ± 1.55 (Fig. 5.2, Appendix F).

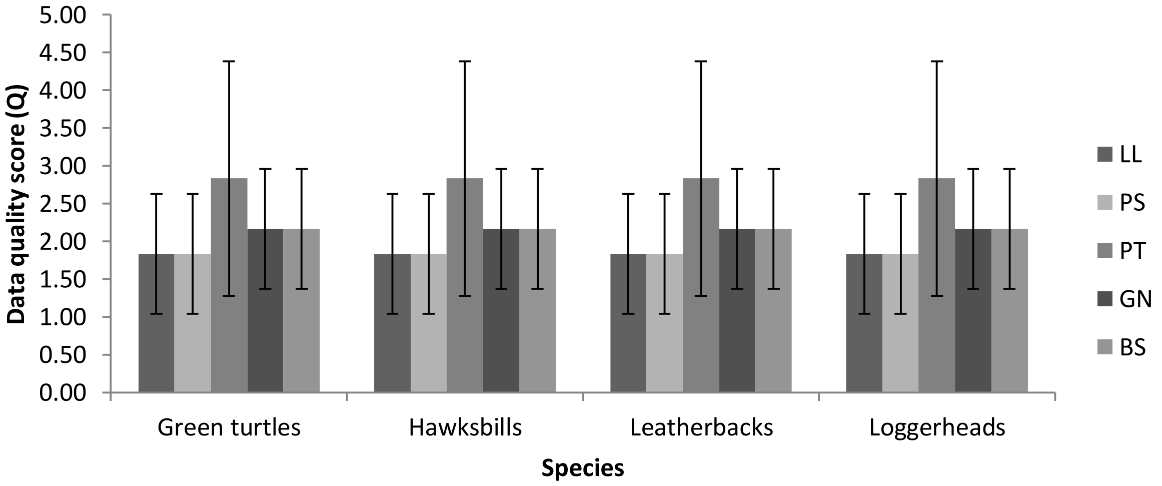


Figure 5.2 Data quality scores (mean ± SD) for the susceptibility analysis of longline (LL), purse seines (PS), prawn trawl (PT), gillnet (GN), and beach seine (BS) fisheries.

Productivity analysis

Data for most of the attributes were available from published literature except for olive ridleys for which very little data are available (Table 5.5). Productivity scores for RMUs ranged between 1.43 ± 0.65 (SD) - 2.50 ± 0.76 (Fig 5.10, Appendix G), with a mean of 2.03. Each of the species was ranked according to their productivity scores (Fig. 5.3). The highest ranked (most productive) species is green turtles with the least productive species being olive ridleys. The confidence in productivity scores are indicated by the data quality scores. There is low confidence in the productivity score of olive ridleys, and this score is likely to bias further results because the low productivity is an artifact of the low data availability.

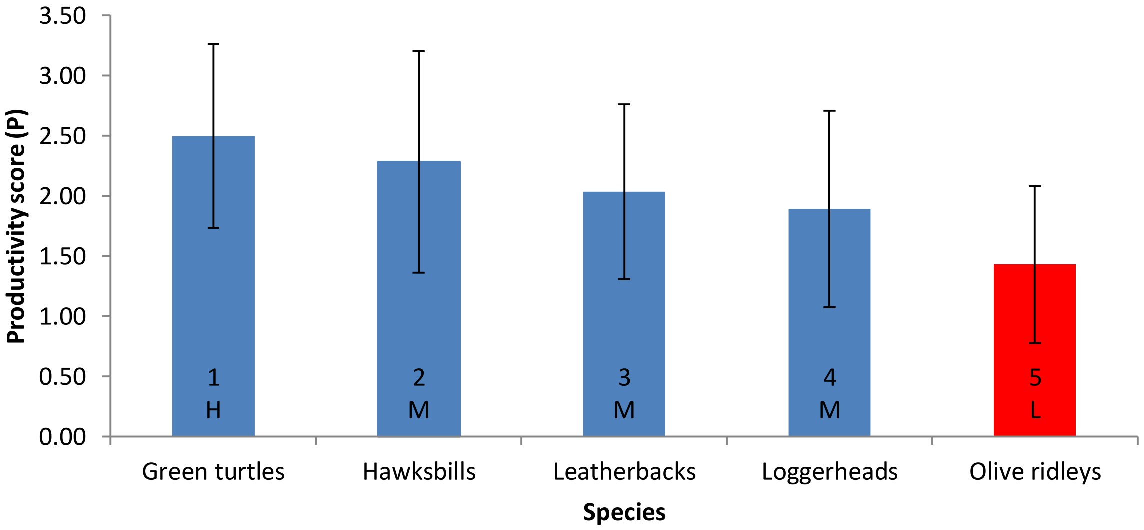


Figure 5.3 Productivity scores of sea turtle RMUs in the SWIO region. Data quality is indicated by different colors (blue = high quality, and red = medium quality). Data labels indicate the productivity ranking of each of the species and whether productivity is regarded as high (H), medium (M), or low (L).

Table 5.5 Population size, trend, and biological data of the five sea turtle RMUs that were used in the productivity assessment.

Species	**Green turtles**	**Hawksbills**	**Leatherbacks**	**Loggerheads**	**Olive ridleys**
Long-term trend	Decline[1-10]	Decline[11-17]	Stable[18]	Increase[18]	Decline[19,20]
Recent trend	Increase[5-10,21,22]	Decline[11,23]	Stable[18]	Increase[18]	Decline[19,20]
Population size	Very large[24]	Very large[24]	Very Small[18,24]	Small[18,24]	Small[24]
Genetic diversity	3[25,26]	DD (1)	1[27]	1[28]	DD
Age at maturity (years)	33[1]	>30[11]	16[29]	36[30]	13[31]
Reproductive lifespan (years)	19.6[1]	21[32]	16[18]	20[18]	DD
Generation length (years)	42.8[1]	41[32]	24[18]	45[18]	DD
Remigration Interval (years)	2.6 - 3[21,33]	2.5[32]	2.9[34]	2.6[35]	DD
Clutches per season	2.7 - 3.5 [21,33]	3.6 - 4[32,36]	6.8[37]	3.7[38,39]	DD
Eggs per clutch	114-124[40,33]	182[36]	104[37]	106[37]	100[31]
Nest success (%)	88[33,40,41]	70 – 86[36,41,42]	78[43]	89[43]	DD
Emergence success (%)	84[33,40,41]	High[41,42]	73[43]	72[43]	DD

References

1. MTSG (2004)
2. Lebeau *et al.* (1979)
3. Le Gall *et al.* (1986)
4. Rene and Roos (1996)
5. Lauret-Stepler *et al.* (2007)
6. Louro *et al.* (2006)
7. Pereira *et al.* (2009)
8. Mortimer *et al.* (2006)
9. Seminoff *et al.* (2004)
10. Seminoff *et al.* (2007)
11. Mortimer and Donnelly (2007)
12. Frazier (1982)
13. Okemwa *et al.* (2004b)
14. Mortimer (1984)
15. Mortimer (2004)
16. Mortimer *et al.* (2006)
17. Howell and Mbindo (1996)
18. Nel *et al.* (2013a)
19. Abreu-Grobois (2006)
20. Plotkin (2007)
21. Bourjea *et al.* (2007a)
22. Mortimer *et al.* (2011b)
23. Mortimer (2007)
24. Appendix A
25. Formia *et al.* (2006)
26. Bourjea *et al.* (2007b)
27. Dutton *et al.* (1999)
28. Shamblin *et al.* (2014)
29. Jones *et al.* (2011)
30. Tucek *et al.* (2014)
31. Zug *et al.* (2006)
32. Mortimer and Bresson (1999)
33. Anastácio *et al.* (2014)
34. Hughes (1996)
35. Hughes (1982)
36. Diamond (1976)
37. Hughes (1974a)
38. Hughes (1974c)
39. Rees *et al.* (2010)
40. Innocenzi *et al.* (2010)
41. Garnier *et al.* (2012)
42. Hitchins *et al.* (2004a)
43. De Wet (2012)

Susceptibility analysis

Leatherbacks (2.17 ± 0.45) are the most susceptible to longline fisheries, with green turtles (1.67 ± 1.09) showing the lowest susceptibility (Fig. 5.4, Appendix G). In purse seine fisheries hawksbills and loggerheads are the most susceptible both with S = 1.67 ± 0.86, with leatherback being the least susceptible (1.33 ± 0.54). Loggerheads (2.00 ± 0.67) show the highest susceptibility to the prawn trawl fishery with green turtles being the least susceptible (1.33 ± 0.86). Loggerheads and leatherbacks are the most susceptible to the gillnet fishery both with S = 2.50 ± 0.88, with hawksbill being the least susceptible (2.00 ± 0.67). Susceptibility to the beach seine fishery is high for loggerheads (2.50 ± 0.88) and hawksbills (2.00 ± 0.67), with leatherbacks being the least susceptible to this fishery. The gillnet fishery has the highest impact across all species (mean susceptibility score of 2.21), followed by beach seine and longline fisheries, with the purse seine fishery having the lowest impact across all species (Fig. 5.5)

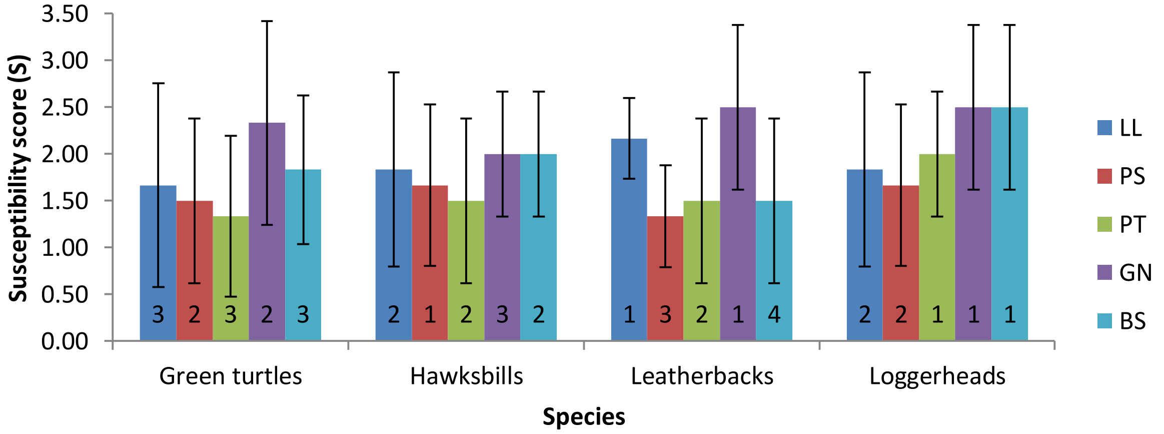


Figure 5.4 Susceptibility scores for scenario 2 of each of the sea turtle RMUs for longline (LL), purse seine (PS), prawn trawl (PT), gillnet (GN) and beach seine (BS) fisheries. Error bars indicate weighted standard deviation of susceptibility scores. Data labels indicate the ranking of each of the species within the corresponding fishery (1 – 5).

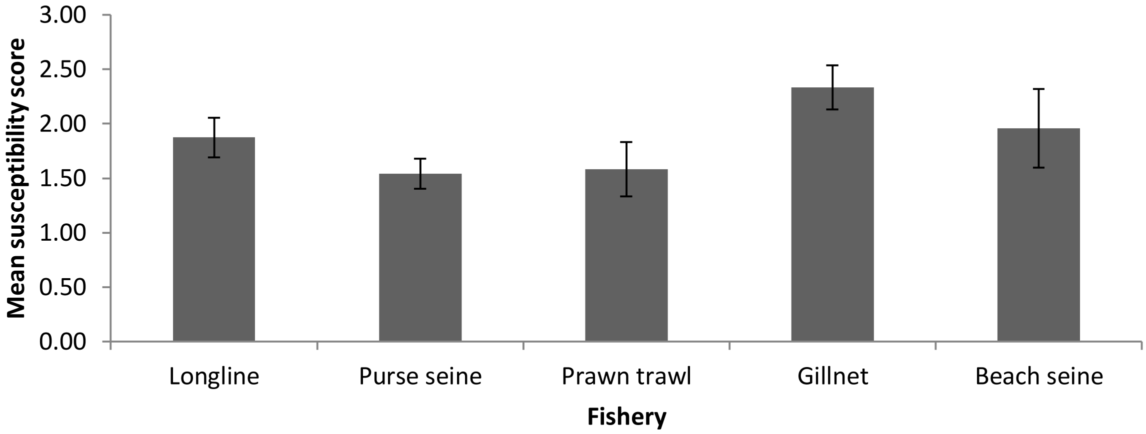


Figure 5.5 Mean susceptibility scores for each of the fisheries assessed. Error bars indicate standard deviation.

Vulnerability analysis

Overall the highest vulnerability was obtained for loggerheads in the beach seine and gillnet fisheries followed by leatherbacks in the gillnet and longline fisheries (Figs. 5.6, 5.7, Table 5.6). The lowest vulnerability was calculated for green turtles in the prawn trawl, purse seine, and longline fisheries. In the longline fishery the species most vulnerable to the fishery is loggerhead turtles and the species least vulnerable is green turtles. The purse seine fishery poses the highest threat to loggerhead turtles and the lowest threat to green turtles. In the prawn trawl fishery, loggerheads show the highest vulnerability with green turtles showing the least vulnerability. The species most vulnerable to gillnet fisheries are loggerheads with the species with the least vulnerability being hawksbill turtles. Loggerhead turtles are the species most vulnerable to beach seine fisheries, with leatherbacks being the least vulnerable to this fishery. Overall, gillnet fisheries pose the greatest threat across all species followed by beach seine and longline fisheries, with purse seine fisheries posing the least threat (Fig 5.8).

Green turtles and hawksbills show the highest vulnerability to gillnet fisheries compared to other fisheries and show the least vulnerability to prawn trawl fisheries (Fig. 5.6 and 5.7). Leatherback turtles are most vulnerable to longline, gillnet fisheries, and are the least vulnerable to purse seine fisheries. Loggerhead turtles are the most vulnerable to beach seine and gillnet fisheries and the least vulnerable to purse seine fisheries.

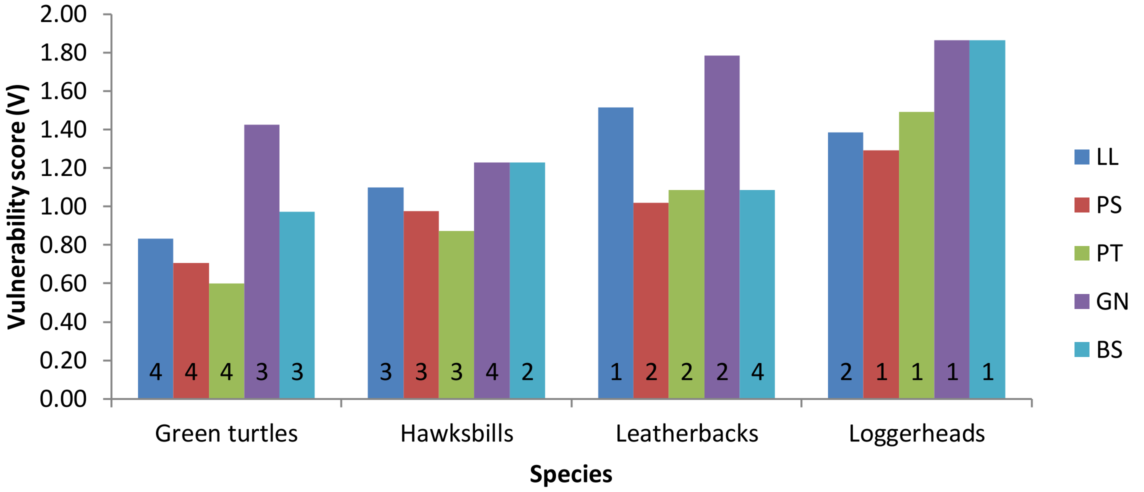


Figure 5.6 Vulnerability scores for each of the species for longline (LL), purse seine (PS), prawn trawl (PT), gillnet (GN), and beach seine (BS) fisheries. Data labels indicate the ranking of each of the species within each of the fisheries.

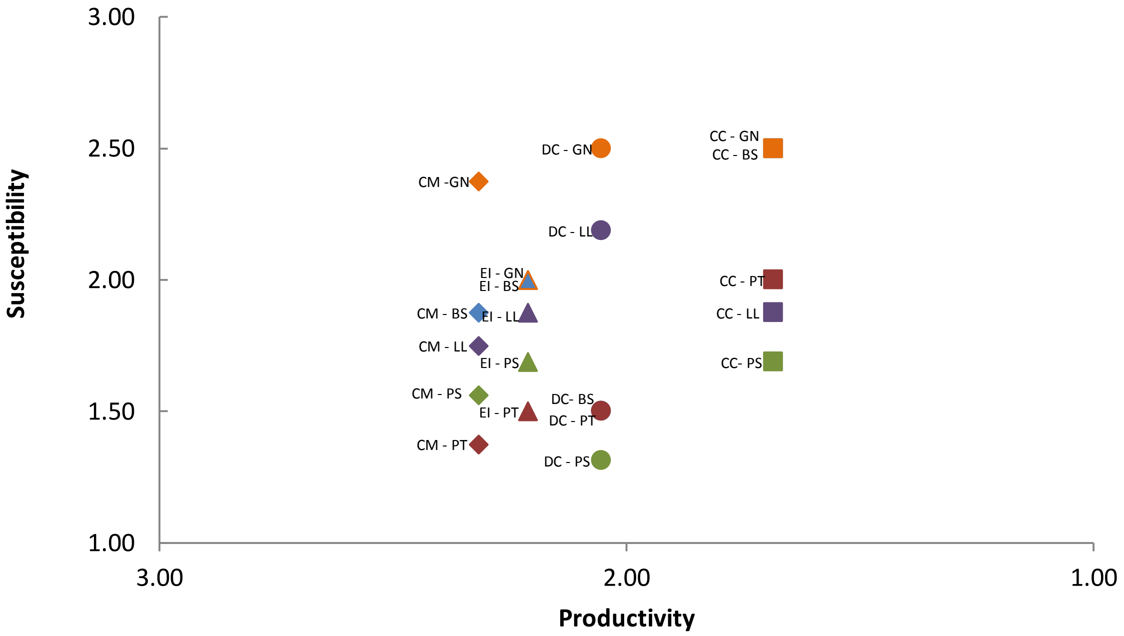


Figure 5.7 Vulnerability plot showing relative vulnerability of sea turtle species (green turtles (CM), hawksbills (EI), leatherbacks (DC), and loggerheads (CC) to longline (LL), purse seine (PS), prawn trawl (PT), gillnet (GN) and beach seine (BS) fisheries in the SWIO.

Table 5.6 Summary of vulnerability scores and rankings for all species in all fisheries including the productivity and susceptibility scores of each. Low vulnerability scores are highlighted with green whilst medium vulnerability scores are highlighted in yellow.

Species	**Fishery**	**Productivity**	**Susceptibility**	**Vulnerability**	**Ranking**
Loggerheads	Beach seine	1.89	2.50	1.86	1
Loggerheads	Gillnet	1.89	2.50	1.86	1
Leatherbacks	Gillnet	2.04	2.50	1.78	2
Leatherbacks	Longline	2.04	2.17	1.51	3
Loggerheads	Trawling	1.89	2.00	1.49	4
Green turtles	Gillnet	2.50	2.33	1.42	5
Loggerheads	Longline	1.89	1.83	1.39	6
Loggerheads	Purse seine	1.89	1.67	1.29	7
Hawksbills	Beach seine	2.29	2.00	1.23	8
Hawksbills	Gillnet	2.29	2.00	1.23	8
Hawksbills	Longline	2.29	1.83	1.10	9
Leatherbacks	Beach seine	2.04	1.50	1.09	10
Leatherbacks	Trawling	2.04	1.50	1.09	10
Leatherbacks	Purse seine	2.04	1.33	1.02	11
Hawksbills	Purse seine	2.29	1.67	0.98	12
Green turtles	Beach seine	2.50	1.83	0.97	13
Hawksbills	Trawling	2.29	1.50	0.87	14
Green turtles	Longline	2.50	1.67	0.83	15
Green turtles	Purse seine	2.50	1.50	0.71	16
Green turtles	Trawling	2.50	1.33	0.60	17

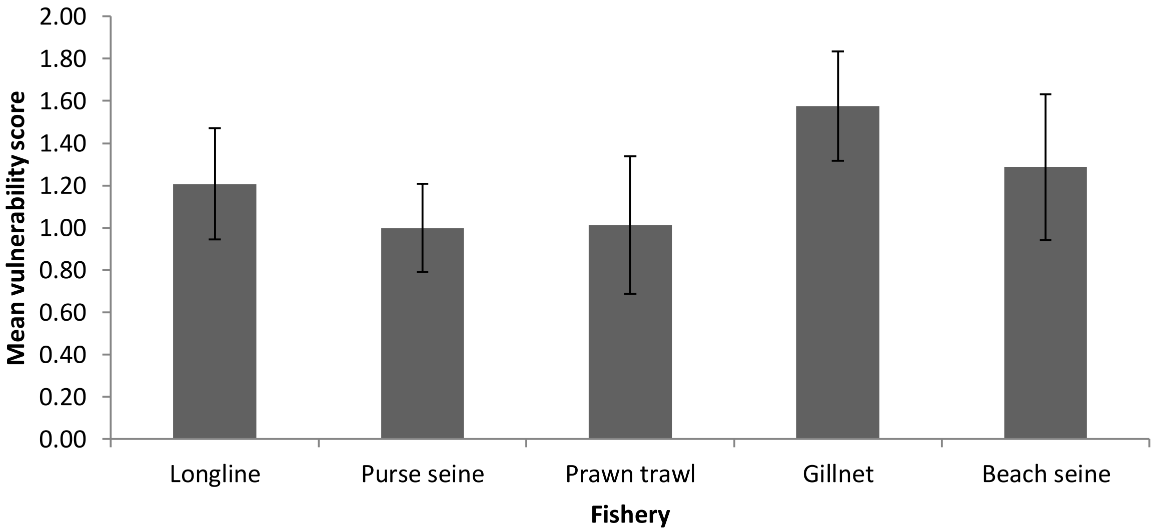


Figure 5.8 Mean vulnerability scores of each of the fisheries. Error bars indicate standard deviation.

Assessment of Cumulative Effects

The cumulative effects assessment of fisheries in isolation of other threats, indicate that loggerheads and leatherbacks are at high risk from fisheries in the SWIO (Fig. 5.9). When the second scenario is assessed where each of the additional threats are included in the analysis together with fishery susceptibility scores it also indicates that loggerheads and leatherbacks are most vulnerable to all impacts that are included in the assessment (Fig 5.10)

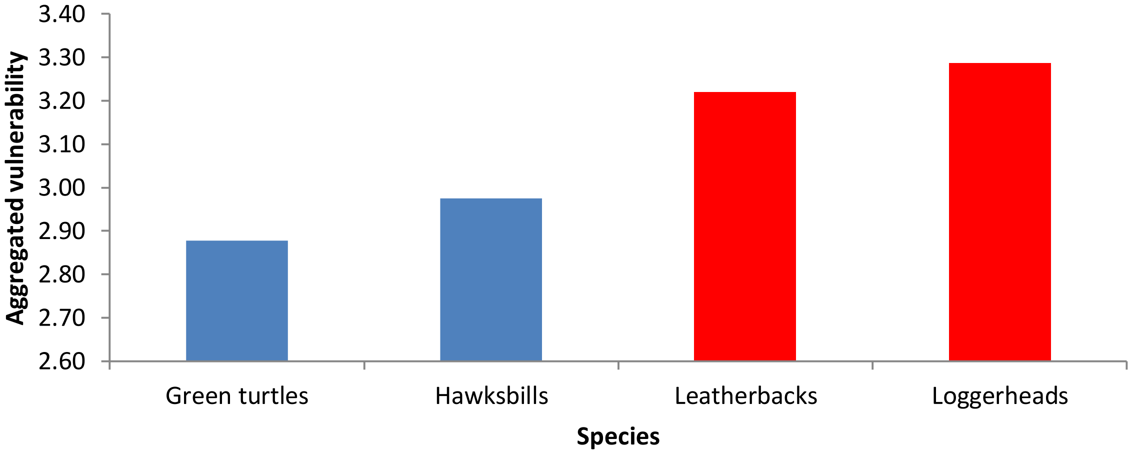


Figure 5.9 Aggregated vulnerability scores of the cumulative effects assessment that were based on only fisheries scores.

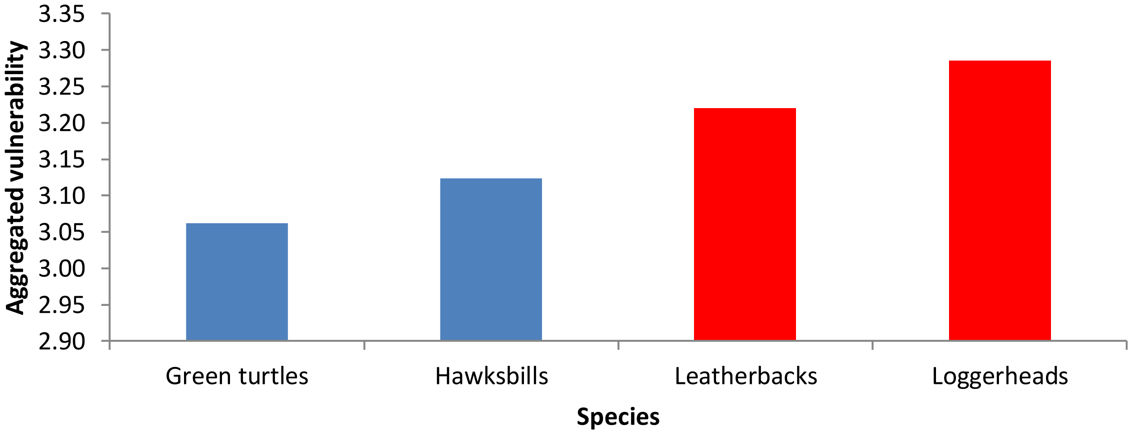


Figure 5.10 Aggregated vulnerability scores of the cumulative effects assessment where the additional threats were treated similarly to fisheries threats in the analysis.

Discussion

In an effort to identify the fisheries responsible for the greatest impact on sea turtle population in the SWIO as well as to identify the species most at risk to fisheries, a PSA analysis, based on the procedure first developed by Milton (2001) and modified for sea turtles by Nel *et al.* (2013b), was performed. The productivity of sea turtle populations and the susceptibility of these populations to fisheries were assessed. A cumulative effects assessment (following Micheli *et al.* (2014)) was performed to identify the species most vulnerable to the host of fisheries in the SWIO. The main results indicated that gillnet fisheries have the greatest relative impact on sea turtle populations compared to other fisheries in the SWIO. Nel *et al.* (2013b) came to the same conclusion when comparing longline, purse seine and gillnet fisheries throughout the Indian Ocean region. The sea turtle RMUs that were found to be most at risk to the combination of fisheries in the SWIO are loggerheads and leatherbacks. This corresponds to a global study by Wallace *et al.* (2011a), that also found that from the sea turtle species in the SWIO region the loggerhead and leatherback population was the most vulnerable to a combination of threats. Loggerheads being the most vulnerable to gillnet and beach seine fisheries and leatherbacks to gillnet fisheries. The vulnerability of loggerheads were surprising considering the fact that this population has been increasing over the past five decades (Nel *et al.*, 2013a), there is however evidence that suggest that substantial overlap occurs between the Arabian Gulf RMU in the north western Indian Ocean region with the SWIO RMU on feeding grounds that occur within the SWIO region (Dalleau *et al.*, 2014). This overlap explains why the loggerhead population been increasing despite the high capture rates that are seen in the gillnet and beach seine fisheries for this species (see Chapter 4 for details), because a high proportion of this bycatch can be assigned to the Arabian Gulf RMU that has ~ 30 000 females nesting annually within the RMU (Nichols, 2007).

Data quality for most of the attributes used in the assessment was acceptable, with the data quality of all productivity and susceptibility analysis being medium or high, except for the olive ridley productivity assessment. Wallace *et al.* (2011a) also found that there are very few biological data available for the species in the WIO region. The low data quality score assessed for the olive ridley RMU in this region is mainly an artefact of the lack of biological data that are available for the species, with data available for only five of the 12 criteria used in the assessment with two of the criteria details being inferred from data for populations outside the region. The low data quality score for this species highlights the need for further research in order to determine the biological criteria for the rookeries in the region. Better data quality will assist in better analysis of the impact that possible threats such as fisheries bycatch can have on this population. Preliminary analysis indicated that this low productivity score results in high vulnerability for this species across all fisheries (Appendix H) PSA graphs). The low data quality for the productivity criteria of this species is thus biasing the results and the species was eliminated from further analysis and reporting. Despite eliminating this species from the analysis, care should still be taken with the management of this species until further assessment with more complete biological data is possible.

There is variation in the resilience of species to impacts; this is evident from the range of productivity scores for the species. Green turtles seem more resilient to impacts than other species as they have the highest productivity score, due to a large population size (Bourjea, 2012) some genetic diversity (Bourjea *et al.*, 2007b), the large size of clutches that females deposit (Anastácio *et al.*, 2014, Innocenzi *et al.*, 2010) and the high hatchling and emergence success (Anastácio *et al.*, 2014, Garnier *et al.*, 2012, Innocenzi *et al.*, 2010). Loggerheads have the lowest productivity score of species in the region indicating that this species will be the least resilient to threats. The low productivity of this species is mainly due to the small population size (Nel *et al.*, 2013a), the low genetic diversity of the RMU (Shamblin *et al.*, 2014), high age to maturity (Tucek *et al.*, 2014) compared to other species, long generation length of the species (Chapter 2), and the low number of clutches that females lay during a nesting season (Hughes, 1974c, Rees *et al.*, 2010).

Loggerheads and leatherbacks are most susceptible to the impact of the gillnet fishery with loggerheads also being susceptible to the beach seine fishery. This supports the findings of De Wet (2012), that found that the artisanal fisheries (including the gillnet and beach seine fisheries) in the SWIO were the biggest threat to the sea turtle populations in South Africa. Both these fisheries are unselective (Hamley, 1975, Tietza *et al.*, 2011), having high capture rates of most species. In the case of leatherbacks and loggerheads, there is larger spatial overlap with these fisheries compared to other species in the region and they have high capture rates in these fisheries relative to their population sizes (see Chapter 4). The lowest susceptibility score was for green turtles in the prawn trawl fishery and leatherbacks in the purse seine and beach seine fisheries. The low susceptibility of these species to the fisheries can in part be ascribed to the low capture rates of the species in the fisheries in relation to their population sizes (see Chapter 4). Despite the fact that green turtles are the species most frequently caught in prawn trawl fisheries in the region (Joynson-Hicks & Ngatunga, 2009, Wamukoya, 1996, West, 2010), the capture rate of this species in the fishery is low compared to the large population size of this species in the region. Leatherbacks on the other hand have a low capture rate in the gillnet fishery (Chapter 4), however the capture rate relative to the population size is high.

All species across all fisheries have either medium vulnerability to fisheries ($V < 1.89$), with low vulnerability ($V < 0.94$) for hawksbills in the prawn trawl fishery and green turtles in the longline, purse seine and prawn trawl fisheries. Leatherbacks show high vulnerability to the gillnet fishery due to the high susceptibility of this species coupled with a relatively low productivity of this species. The capture rates of leatherbacks in the gillnet fishery are high relative to the population size of this species (Chapter 4). Leatherbacks however are not the least productive species in the region, with females depositing more clutches per season than for any other species (see Chapter 2, Table 5.5), coupled with high nest and emergence success (De Wet, 2012), and lower age to maturity (Jones *et al.*, 2011) increasing their productivity. However, the population size is small (<100 nesters annually, Nel *et al.*, 2013a), exhibiting a short reproductive lifespan (Nel *et al.*, 2013a), and they have low genetic diversity (Dutton *et al.*, 1999), decreasing the productivity of the species.

Loggerheads compared to other species show relatively higher vulnerability to most of the fisheries than other species. The highest vulnerability for this species is seen in the beach seine and gillnet fisheries. This is due to the low productivity of the species that is largely attributed to the small population size (< 1000 annual nesters, Nel *et al.*, 2013a), low genetic diversity (Shamblin *et al.*, 2014), high age to maturity (Tucek *et al.*, 2014), and females producing fewer clutches per season than other species (see Chapter 2 for review, Table 5.5). The population of loggerheads in the SWIO has increased in recent years, thus this high vulnerability may be due to the overlap with the Arabian Gulf RMU on feeding grounds as explained above. Green turtles on the other hand show the least vulnerability to most of the fisheries. This is mainly attributable to the fact that green turtles are the most productive species in the region (as explained above). The capture rates of this species in most of the fisheries are relatively low compared to the population size (see Chapter 3 and 4), except for capture in the gillnet fishery that exceeds the population size (see Chapter 4).

The PSA indicated that gillnet fisheries are likely to have the greatest impact on sea turtle populations in the region with the highest average vulnerability score; 1.58 ± 0.26 (SD). This fishery has extremely high capture rates compared to the other fisheries (see Chapter 3 and 4), with high mortality of captured individuals associated with this fishery (Humber *et al.*, 2011, Kiszka, 2012a), thus posing a significant threat to the sea turtles. Purse seine (excluding the impacts of FADs) and prawn trawl fisheries (excluding with the use of TEDs) are likely to have the smallest impact on sea turtles populations compared to other fisheries; they had the lowest average vulnerability scores (of 1.00 ± 0.21 (SD) and 1.01 ± 0.33 (SD), respectively). Both the purse seine and prawn trawl fisheries in the SWIO region have low annual capture rates of sea turtles (see Chapter 3 and 4), thus decreasing the threat that these fisheries pose to populations in the region. However, it should be noted that the low capture rate in the prawn trawl fishery in the region is a reflection of the low effort that is currently exerted in the region. Historically much higher effort (Fennessy, 2012) was exerted throughout the region, however there has been a substantial decline in effort especially in South Africa, Kenya and Tanzania (Fennessy, 2012), with the declines in effort due to closure of the fishery in Tanzania (WIOFISH, 2011), and a decrease in target catches and thus the economic viability of the fisheries in both South Africa and Kenya (Fennessy, 2012). Should there be an increase in the effort of the prawn trawl fishery in the region it is expected that this fishery will pose a much higher risk to sea turtles than what is currently the case. A decrease in effort in the prawn trawl fishery in South Africa has possibly led to an increase in the number of females nesting per annum in the region (De Wet, 2012). It is expected that should the situation be reversed, there will be an increase in the impact of the fishery on the sea turtle populations and thus a subsequent slowing of the growth rates of the populations.

Loggerheads and leatherback are shown to have the highest vulnerability to the combination of fisheries. This is evident from the high aggregated vulnerability scores for these two species when cumulative effects of fisheries are investigated in isolation of other threats. The population sizes for both these species in the region is small (Nel *et al.*, 2013a), thus the cumulative effect of bycatch in fisheries will have a larger impact on these two populations than on those with relatively larger population sizes. However, the loggerhead population has been increasing over several decades whereas the leatherback population in South Africa has been stable. There are several possible reasons for this difference. The analysis focused mainly on fisheries within the SWIO region, and does not cover fishery-related mortality in the Atlantic Ocean. Both these species have been shown to frequent the waters of the

Atlantic Ocean (Luschi *et al.*, 2006). Thus, a possible explanation for this discrepancy in results can be that the leatherback population is facing a higher impact in the Atlantic Ocean region compared to loggerhead turtles. Another possible explanation is that the loggerhead bycatch in the SWIO is overestimated. There is significant overlap between the populations that nest in the SWIO with the Arabian Gulf population at feeding grounds (as explained above). This overlap is likely to be greater in the upper part of the SWIO in areas such as Tanzania and Kenya due to the proximity of these areas to other rookeries. High catch rates are also assessed for loggerheads in this region (Chapter 4).

When considering the cumulative effects of fisheries combined with alternative threats, loggerhead and leatherback populations are shown to be at greatest risk to the combination of threats; this is due to their small population sizes as mentioned above. These results highlight the fact that even though small numbers of a species may be captured in a fishery (such as the small number of mortalities for leatherbacks in both gillnet and longline fisheries), these fisheries may have a significant threat on the population in combination with other threats. This will be particularly true for small populations such as the leatherback population in the SWIO.

Mitigation of bycatches has focused mainly on industrial fisheries such as longline (Gilman *et al.*, 2006a) and gillnet fisheries (Gilman *et al.*, 2010a), mostly because it poses less of a challenge than mitigating the impacts of artisanal fisheries. Artisanal fisheries often lack management, and when such management exists there is frequently a lack of enforcement due to the nature of the fisheries and the remoteness of landing sites (Kimani *et al.*, 2009). The potentially higher impact of gillnet fishery on sea turtle populations makes it necessary to focus more attention on reducing bycatch and mortality in this fishery, despite the difficulty that is faced due to the artisanal nature of this fishery. It however does not mean that when a fishery is expected to have a lower impact on sea turtles that management focus should move away from those fisheries. The existing management might already be the reasons for lower vulnerability of species in these fisheries, but it is still necessary to reduce the number of mortalities of these endangered animals from every threat.

Despite the fact that this assessment included a host of fisheries (known to impact on sea turtle populations worldwide), there are several alternative threats (e.g. boat strikes and entanglement in FADs, dynamite fishing; (Bourjea *et al.*, 2014, De Wet, 2012, West, 2010) that have not been included in these assessments, which may be responsible for the slow recovery of populations in the SWIO. No data were available for the size classes that are caught in fisheries, this may be an additional factor that has a significant influence on the impact that a fishery has on sea turtle populations (Crouse *et al.*, 1987, Heppell, 1998), removing older age classes that have higher reproductive value can be detrimental to a populations growth rate (Heppell, 1998). In order for more thorough assessment in the future it is necessary to have specifics of age classes that are caught in fisheries to better assess the relative impact. There is also no information currently available on the sex ratio between breeding males and females at nesting beaches. This ratio is most likely to be skewed towards females (Tomillo *et al.*, 2014), which may have an effect on the mating system and thus the productivity of the species (Stewart & Dutton, 2014).

The analysis is overall robust to determine the relative risks that fisheries pose to sea turtles populations. However, the analysis can be improved with more complete data from fisheries as well as biological and survivorship data from local populations. More in depth analysis will be possible when especially with more complete data on bycatches and fishing effort (especially for the artisanal fisheries) in the region. More complete bycatch data, including areas of higher captures and seasonal differences in captures, will assist in identifying regional hotspots of bycatch. Better data on the densities of sea turtles within their distributions will allow for the identification of areas where there is higher probabilities of interaction, such as areas with high sea turtle densities that overlap with areas of high fishing effort.

The risk posed by fisheries to sea turtle populations in the SWIO is generally greater than those posed by land-based pressures because most of the populations have at least some part of their nesting area protected. The distinction between the relative risks of fisheries however does not indicate that less attention should be given to fisheries that pose a smaller risk. Sea turtle populations are endangered and reducing the mortality from all sources remains an important factor for effective conservation of these species.

Chapter 6
Conclusion

The extent to which sea turtle populations have been protected historically throughout the SWIO varies, however recently at least part of each of the populations have received significant protection that has led to an increase in the population number of green turtles and loggerheads (Mortimer *et al.*, 2011b, Nel *et al.*, 2013a). The hawksbill and olive ridley populations in the region are however still showing a decline in population numbers with the population size of leatherback showing a stable population trend. The loggerhead and leatherback populations in the SWIO have received similar protection at nesting beaches, however these two populations have not responded equally indicating that protection measures at nesting beaches may not in all cases be sufficient to facilitate the recovery of populations (Nel *et al.*, 2013a). This necessitates looking at the mortality that affects these populations in their oceanic environments. Despite the fact that sea turtles face several threats in their oceanic environment, very little information is available regarding the magnitude of the impact that these threats have on these populations. This Dissertation focused on assessing the relative impact that fisheries have on sea turtle populations because fisheries are regarded as having the greatest impact on populations globally (Lewison & Crowder, 2007, Wallace *et al.*, 2011a).

This study showed that, of the industrial fisheries investigated, longline fisheries have a much higher impact on sea turtle populations than purse seine fisheries. This corresponds to studies in other regions where purse seine fisheries also have substantially lower impacts (Angel *et al.*, 2014). The longline fishery not only has higher bycatch rates compared to the purse seine fishery but also has higher mortality rates (indiv.y^{-1}). The longline fishery mostly captures loggerhead and leatherback turtles similar to longline fisheries in other regions (Carreras *et al.*, 2004, Lewison *et al.*, 2004b, Polovina *et al.*, 2003a), whereas the purse seine fishery most commonly captures olive ridleys, hawksbills and green turtles. Similar patterns are observed in purse seine fisheries in Eastern Pacific (Hall & Roman, 2013) and Western Pacific (OFP, 2001), however in the Atlantic Ocean, Amandè *et al.* (2010) found green turtles (30 %), kemps ridleys and leatherbacks (~ 17 % for both species) to be the most commonly caught species, with olive ridleys being the least common species caught. Another study in the Atlantic Ocean determined olive ridleys and loggerheads to be the most commonly caught species (Clermont *et al.*, 2012). The difference in species composition of catches can be attributed to spatial overlap with fisheries and the relative abundances of the species (Clermont *et al.*, 2012).

Longline fisheries have the greatest impact on the leatherback population because the annual number of mortalities in this fishery equates to approximately 79 % of the annual number of nesting females, whereas the mortalities for other species are considerably lower. The longline fishery in the region may be one responsible for slowing of the recovery of this species in the SWIO; however, this is not the case for any of the other species. In other ocean regions leatherback turtles are also seen as the species most impacted by the longline industry (Lewison *et al.*, 2004b). The annual number of sea turtles captures and mortalities per species in the purse seine fishery is considerably lower than the annual number of nesting females and it is thus not expected that this fishery is responsible for the slowing of population growth rates in the SWIO region.

The magnitude of captures of the different species in the region is not explained by the relative abundances of the species, as there is no correlation between the relative abundances and the bycatch of species. However, the bycatch in the purse seine fishery is correlated with the percentage horizontal overlap with the different species, with bycatch increasing with an increase in the horizontal overlap. For the longline fishery, it is thus expected that the difference in magnitude of bycatch of the species be related to factors involving the behaviour and habitat use of the species. For the purse seine fishery the differences in magnitude of bycatch is also not completely explained by the horizontal overlap and it is expected that the difference may be related to a combination of the differences in relative abundance and the behaviour and habitat choice of species in combination with the overlap.

Of the coastal fisheries investigated, the gillnet fishery has the highest annual bycatch of sea turtles in the region followed by beach seine and prawn trawl fisheries. The captures of gillnet fisheries amount to approximately 77 % of the 52 370 sea turtles that are captured in the coastal fisheries in the region. The captures in coastal fisheries in several regions also indicate that gillnet fisheries have higher capture rates per year than prawn trawl fisheries (Wallace *et al.*, 2010b), however in the Gulf of Mexico (Finkbeiner *et al.*, 2011), Mediterranean (Casale, 2011), and USA (Moore *et al.*, 2009), trawl fisheries were responsible for higher capture rates of sea turtles than gillnet fisheries. The difference in capture rates within the fisheries with other regions may be ascribed to differences in the effort exerted as well as the differences in densities of sea turtles within the fished areas. Within the region a greater effort is exerted in the gillnet fishery than in the beach seine and prawn trawl fisheries.

The species most commonly captured in the gillnet fishery in the SWIO is green turtles, this is similar to Peru (Alfaro Shigueto *et al.*, 2011, Alfaro Shigueto *et al.*, 2010), and Mexico (Mancini *et al.*, 2011), however in the US mid-Atlantic fleet, loggerheads were the most commonly captured species (Murray, 2009), and in the West Indies in Trinidad and Tobago leatherbacks were the most commonly caught species (Lum, 2006). In the prawn trawl fishery in the SWIO, the most commonly captured species were green turtles followed by loggerheads and hawksbills. In the Northern prawn trawl fishery in Australia, the highest capture rates were for olive ridleys and green turtles (Brewer *et al.*, 2006), and in the Queensland trawl fishery in Australia green turtles were also the most commonly caught species (Robins, 1995). The difference in species composition of captures in different regions may be partially ascribed to the differences of abundances of the species in the regions. With the high annual capture and mortality rates of green turtles in the gillnet fishery in the region it is expected that this fishery might be responsible for slowing the recovery rate of this species in the region. However, the other fisheries are not expected to have a significant impact on the recovery rate of species in isolation of other fisheries.

Results from the PSA assessment indicate that all species have relatively low or medium vulnerabilities to fisheries in the region when these fisheries are considered in isolation. Gillnet fisheries have the highest average vulnerability score indicating that this fishery is likely to have the greatest impact on sea turtle populations in the region. However, in a situation where multiple fisheries are combined with other threats impact upon populations it is necessary to assess the cumulative impact that these have on sea turtle populations. Loggerheads and leatherbacks are shown to be the most vulnerable to the combination of threats that are exerted; however, the loggerhead population has shown increase in population numbers over several decades.

Despite the fact that this study is the most comprehensive study on the bycatch of sea turtles in the SWIO region there are several limitations and data gaps that were highlighted during the study. The limited data available regarding bycatch rates for all fisheries in the region forced simplistic analysis; however, the analysis is still robust enough to identify the relative impact that the fisheries will have on sea turtle species. Mostly, even distributions of sea turtle species were assumed due to a lack of data on the seasonal distribution and density hotspots of these species in the region. Available satellite tracking data for the different species however does allow for the analysis of such hotspots. The raw data for this type of analysis is however not easily accessible due to limited sharing by data owners. With the necessary available data, hotspots of sea turtle densities can be identified as well as high use areas of each species in the region that can assist in focussing mitigation measures such as time area closure on areas where a higher risk is perceived. Effort data recording from fisheries are often not detailed enough on large databases such as the IOTC database and separation between different gear configurations is often not possible. Should better recording of data be done, it will be possible to refine the bycatch estimates to give a clearer indication as to where mitigation measures are most needed. Reliable estimates of post-release mortality are not available for fisheries in the SWIO and these differ greatly amongst other regions as well, making it impossible to apply the statistics to this region. Post-release mortality however remains an important factor to consider when doing bycatch assessments. This is because a significant portion of the actual mortality due to fishing might not be assessed without reliable post-release mortality rates.

Several mitigation measures (Appendix C) do exist that have the potential to decrease bycatch in both artisanal and industrial fisheries, however there remains a substantial concern for the implementation of these measures especially in the artisanal fisheries where monitoring and enforcement capabilities are often lacking or absent. Due to this, it is imperative to attempt to reduce bycatch related mortality of those species most at risk to specific fisheries to ensure the persistence of these species in the region. Mitigation in the artisanal fisheries pose significant difficulties due to the value that these fishers place on sea turtles as a source of nutrition, medicinal purposes or as a source of income. Investigation of alternative sources of livelihoods and nutrition may assist in the reduction of threat to the species as is evident from the turtle monitoring program in South Africa that offers employment opportunities for the local community that aids in the conservation of species.

Appendix A
Nesting abundances

Table A.1 Green turtle nesting abundances for rookeries in the SWIO. Details are given per site in each country where nesting of green turtles has been identified. Sites with "zero" females are sites where nesting activity takes place but where the number of females nesting per year has not been quantified. All data was extracted from the SWOT online turtle database (Kot *et al.*, 2013) unless otherwise noted.

Country	Site	Year	Females
Comoros	Fomboni-Hoani	2010	64
Comoros	Moheli	2005	211
French Scattered Islands	Europa	2009	2126
French Scattered Islands	Glorieuses	2009	1114
French Scattered Islands	Isles Eparses	1996	0
French Scattered Islands	Juan de Nova	2009	4
French Scattered Islands	Tromelin	2009	595
Kenya	Chandani	2007	5
Kenya	Kitanga kikuu	2007	3
Kenya	Kiunga	2007	3
Kenya	Kiwaiyu	2007	2
Kenya	Kongowale	2007	3
Kenya	KSV	2007	1
Kenya	Kui	2007	1
Kenya	Magogo	2007	5
Kenya	Mongoni	2007	3
Kenya	Mtumumwe	2007	2
Kenya	Mwanabule	2007	2
Kenya	Mwongo shariff	2007	3
Kenya	Porcupine	2007	1
Kenya	Watamu	2000	1
Madagascar	Barrens Archipelago	2005	0
Madagascar	Nosy Iranja Kely	2003	35
Madagascar	Tulear	1999	300
Mayotte	Comoros Archipelago	2007	0
Mayotte	Grande Saziley Beach	2005	399
Mayotte	Maoussi, Majicavo 1, 2, 3 and 4	2005	1278
Mozambique	Bazaruto National Park	2009	1
Mozambique	Primeiras and Segundas Islands	2004	0
Mozambique	Quirimbas National Park	2008	1
Mozambique	Rongui Island and Vamizi Island	2008	63
Réunion	All island	2009	0
Réunion	Réunion Island	2006	1
Seychelles	Aldabra Islands	2007	0
Seychelles	Anse Barbarons, West Mahe	2004	1
Seychelles	Anse Bonne Care, Pezula Resort	2006	0
Seychelles	Anse Cachee	2009	1
Seychelles	Anse Forbans	2008	0
Seychelles	Anse Georgette	2007	2
Seychelles	Anse Grand Police	2009	1
Seychelles	Anse Intendance	2009	2
Seychelles	Anse Louis, West Mahe	2008	0

Country	**Site**	**Year**	**Females**
Seychelles	Anse Parnell	2008	0
Seychelles	Anse Petite Boileau	2009	1
Seychelles	Anse Petite Marie Louise, South East Mahe	2008	0
Seychelles	Anse Riviere Gaspart, West Mahe	2008	0
Seychelles	Anse Soleil, West Mahe	2006	0
Seychelles	Anse Takamaka	2008	0
Seychelles	Assumption Island	2007	0
Seychelles	Baie Lazare, West Mahe	2004	0
Seychelles	Bijoutier Island	1984	55
Seychelles	Coetivy Island	1984	55
Seychelles	Cousin Island Special Reserve	2007	1
Seychelles	Cousine Island	2007	13
Seychelles	Farquhar Group	2002	4145
Seychelles	Frigate Island	2010	0
Seychelles	Grande Anse Kerlan	2008	0
Seychelles	Jetty Beach, Pezula Resort	2006	0
Seychelles	Petite Anse	2008	0
Seychelles	Petite Anse Kerlan	2008	0
Seychelles	Petite Anse Soleil, West Mahe	2006	0
Seychelles	Petite Marie Louise	2008	1
Seychelles	Poivre Islands	2010	0
Seychelles	President Villa	2007	0
Seychelles	Presidents Beach, Pezula Resort	2006	0
Seychelles	South Mahe Beach No. 6	2007	2
Seychelles	South Mahe beaches	2008	2
Seychelles	South-western end (Madame Zabre to Bombay)	2004	3
Seychelles	Turtle Beach, Pezula Resort	2006	0
Seychelles	Unamed beach, South East Mahe	2008	0
Seychelles	Villa Beach, Pezula Resort	2006	0
Tanzania	Mafia Island	2004	51
Tanzania	Misali Island, Pembe	2002	9
Tanzania	Mnemba Island, Unguja	2002	12
Tanzania	Mtwara	2004	15
Total			**10599**

Table A.2 Hawksbill turtle nesting abundances for rookeries in the SWIO. Details are given per site in each country where nesting of hawksbill turtles has been identified. Sites with "zero" females are sites where nesting activity takes place but where the number of females nesting per year has not been quantified. All data was extracted from the SWOT online turtle database (Kot *et al.*, 2013) unless otherwise noted.

Country	**Site**	**Year**	**Females**
British Indian Ocean Territory	Diego Garcia	2006	105
British Indian Ocean Territory	Peros Banhos	2006	105
Comoros	Anjouan and Grand Comore	2000	25[1]
Comoros	Moheli	2000	25[1]
French Southern Territories	Juan de Nova	2009	10
French Southern Territories	Glorieuses	2006	<10[1]
Kenya	Kongowale	2006	1
Madagascar	Nosy Iranja Kely	2003	8
Madagascar	Total	2001	1000[1]
Mayotte	Mayotte Island	2006	9
Mozambique	Paquissico Tchawane Beach and Lemani Beach	2005	
Mozambique	Quirimbas National Park	2008	1
Mozambique	Rongui Island and Vamizi Island	2008	1
Seychelles	Aldabra Islands	2005	18
Seychelles	Amirantes	2010	
Seychelles	Anse Barbarons, West Mahe	2007	1
Seychelles	Anse Bazarca	2009	11
Seychelles	Anse Bonne Care, Pezula Resort	2006	0
Seychelles	Anse Bougainville	2009	1
Seychelles	Anse Cachee	2009	12
Seychelles	Anse Capuchin	2009	1
Seychelles	Anse Corail	2009	5
Seychelles	Anse Forbans	2009	1
Seychelles	Anse Georgette	2009	1
Seychelles	Anse Gouvernment	2009	3
Seychelles	Anse Grand Police	2009	15
Seychelles	Anse Intendance	2009	6
Seychelles	Anse Lascars	2009	3
Seychelles	Anse Louis	2009	1
Seychelles	Anse Parnell	2009	2
Seychelles	Anse Patates	2008	2
Seychelles	Anse Petite Boileau	2009	7
Seychelles	Anse Petite Marie Louise, South East Mahe	2008	2
Seychelles	Anse Petite Police	2009	5
Seychelles	Anse Riviere Gaspart, West Mahe	2008	2
Seychelles	Anse Royale	2009	1
Seychelles	Anse Soleil, West Mahe	2006	0
Seychelles	Anse Takamaka	2009	3
Seychelles	Aride Island Nature Reserve	2005	16
Seychelles	Baie Cipailles	2009	1
Seychelles	Baie Lazare, West Mahe	2004	1
Seychelles	Beaches of South Mahe Island	2005	68
Seychelles	Bird Island	2005	49
Seychelles	Cerf Island	2009	1
Seychelles	Cousin Island Special Reserve	2007	793
Seychelles	Cousine Island	2008	16
Seychelles	D'Arros Island and St. Joseph Atoll	2005	375

Country	**Site**	**Year**	**Females**
Seychelles	Eight beaches at Curieuse Island Marine Park	2005	63
Seychelles	Fairy Land	2009	1
Seychelles	Five beaches in the Sainte Anne Marine Park	2005	49
Seychelles	Frigate Island	2005	50
Seychelles	Grande Anse Kerlan	2009	7
Seychelles	Grande Barbe	2009	21
Seychelles	Granitics (point placed on Bird Island)	2010	
Seychelles	Ile Plate	2010	
Seychelles	Jetty Beach, Pezula Resort	2006	1
Seychelles	La Passe	2009	1
Seychelles	Mme Toians	2009	1
Seychelles	NE Point	2009	0
Seychelles	North Island	2005	18
Seychelles	Petite Anse	2008	2
Seychelles	Petite Anse Kerlan	2009	1
Seychelles	Petite Anse Soleil, West Mahe	2006	0
Seychelles	Petite Marie Louise	2008	2
Seychelles	Pointe Etienne	2007	2
Seychelles	President Villa	2009	1
Seychelles	Presidents Beach, Pezula Resort	2006	0
Seychelles	South Mahe beaches	2008	72
Seychelles	South-western end (Madame Zabre to Bombay)	2004	12
Seychelles	Turtle Beach, Pezula Resort	2006	3
Seychelles	Unamed beach, South East Mahe	2008	0
Seychelles	Villa Beach, Pezula Resort	2006	1
Tanzania	Mafia Island	2006	1
Total			**2762**

References
1. Mortimer and Donnelly (2007)

Table A.3 Leatherback turtle nesting abundances for rookeries in the SWIO. Details are given per site in each country where nesting of leatherback turtles has been identified. All data was extracted from the SWOT online turtle database (Kot *et al.*, 2013) unless otherwise noted.

Country	**Site**	**Year**	**Females**
Mozambique	Bazaruto National Park	2009	1
Mozambique	Bilene Beach	2009	2
Mozambique	Inhaca Island	2009	2
Mozambique	Malongane	2009	10
Mozambique	Manhica	2008	2
Mozambique	Milibangalala Beach	2008	2
Mozambique	Mucombo-Santa Maria	2009	2
Mozambique	Ponta de Ouro	2009	2
South Africa	Mabibi to Kosi Lake	2005	49
Total			**72**

Table A.4 Loggerhead turtle nesting abundances for rookeries in the SWIO. Details are given per site in each country where nesting has been identified. Sites with "zero" females are sites where nesting activity takes place but where the number of females nesting per year has not been quantified. All data was extracted from the SWOT online turtle database (Kot *et al.*, 2013) unless otherwise noted.

Country	Site ID	Year	Females
Madagascar	Beaches around Besambay and Maromena	2004	
Madagascar	Beaches between Fort-Dauphin and Manantenina	2004	16
Mozambique	Bazaruto National Park	2009	1
Mozambique	Bilene Beach	2009	2
Mozambique	Dubela-Mucombo	2009	43
Mozambique	Inhaca Island	2009	6
Mozambique	Malongane	2009	84
Mozambique	Manhica	2009	1
Mozambique	Milibangalala Beach	2008	37
Mozambique	Mucombo-Santa Maria	2009	27
Mozambique	Ponta de Ouro	2009	7
Mozambique	Tofo Beach	2008	2
Mozambique	Xai-Xai Beach	2008	1
Mozambique	Zavala	2009	3
Mozambique	Zavora	2009	2
South Africa	Mabibi to Kosi Lake	2005	362
Total			**594**

Table A.5 Olive ridley nesting abundances for rookeries in the SWIO. Details are given per site in each country where nesting has been identified. All data was extracted from the SWOT online turtle database (Kot *et al.*, 2013) unless otherwise noted.

Country	Site	Year	Females
Kenya	Kiunga	2000	3
Kenya	Mombasa	2000	4
Kenya	Watamu	2000	2
Mozambique	Bazaruto National Park	2008	1
Total			**10**

Table A.6 Olive ridley nesting abundances for rookeries in the WIO, excluding those for population in the SWIO region. Details are given per site in each country where nesting has been identified. Sites with "zero" females are sites where nesting activity takes place but where the number of females nesting per year has not yet been quantified. All data was extracted from the SWOT online turtle database (Kot *et al.*, 2013) unless otherwise noted.

Country	Site ID	Females
India	01 Dakshina Kannada District	0
India	01 Utarra Kannada District	0
India	02 Dakshina Kannada District	0
India	02 Utarra Kannada District	0
India	03 Dakshina Kannada District	0
India	03 Utarra Kannada District	0
India	04 Dakshina Kannada District	0
India	04 Utarra Kannada District	0
India	05 Dakshina Kannada District	0
India	05 Utarra Kannada District	0
India	06 Dakshina Kannada District	0
India	06 Utarra Kannada District	0
India	07 Dakshina Kannada District	0
India	08 Dakshina Kannada District	0
India	09 Dakshina Kannada District	0
India	10 Dakshina Kannada District	0
India	11 Dakshina Kannada District	0
India	12 Dakshina Kannada District	0
India	13 Dakshina Kannada District	0
India	14 Dakshina Kannada District	0
India	15 Dakshina Kannada District	0
India	Achara, Sindhudurg	0
India	Adri-Navapara, Junagadh	3
India	Agatti Island, Laccadive Island Group	8
India	Agonda	43
India	Ambolgad, Ratnargiri	0
India	Amindivi Group	6
India	Amreli	2
India	Andrott Island, Laccadive Island Group	3
India	Anjunem	0
India	Bada-Layja Nana, Kachchh	10
India	Baidher Island, Jamnagar	15
India	Bambhdai-Bada, Kachchh	4
India	Betul	0
India	Bhavnagar	4
India	Bogmalo	0
India	Calanguite	0
India	Dahanu, Thane	0
India	Digha/Dadanpatrabar, Medinipore	49
India	Galgibaga	7
India	Gundilai-Tragadi, Kachchh	2
India	Jambudwip, Sunderban Biosphere Reserve, South 24 Parganas	11
India	Kadmat Island, Amindivi Island Group	3
India	Kalpeni Island, Laccadive Island Group	3
India	Kalpitti Islet, Laccadive Island Group	22
India	Kamond-Suthri, Kachchh	1
India	Kanniyakumari to Tiruchendur 70 km	96
India	Kasarakod	14

Country	Site ID	Females
India	Kashid, Raigad	0
India	Kavaratti Island, Laccadive Island Group	3
India	Kerim	0
India	Kharakhetar-Kuranga, Jamnagar	5
India	Kovalum	0
India	Kozhikode	9
India	Laccadive Group	69
India	Lamba, Jamnagar	3
India	Lamba-Miyani, Jamnagar	1
India	Layja Nana-Mandvi, Kachchh	10
India	Malvan, Sindhudurg	0
India	Mangrol-Bada, Junagadh	3
India	Minicoy Group	1
India	Minicoy Island, Minicoy Island Group	1
India	Mojap-Sivrajpur, Jamnagar	1
India	Morjim	0
India	Mumbai	0
India	Navdra-Lamba, Jamnagar	1
India	Neevati, Sindhudurg	0
India	Palghar	0
India	Porbandhar	65
India	Rahij-Maktupur, Junagadh	4
India	Ratnagiri, Ratnagiri	0
India	Redi, Sindhudurg	0
India	Shill-Lohej, Junagadh	1
India	Shiroda-Aravali, Sindhudurg	0
India	Srikakulam	129
India	Srivardhan, Raigad	0
India	Suheli Cheriyakara, Laccadive Island Group	22
India	Utorda	0
India	Velas	7
India	Velneshwar, Ratnagiri	0
India	Velye, Ratnagiri	0
Kenya	Kiunga	3
Kenya	Mombasa	4
Kenya	Watamu	2
Mozambique	Bazaruto National Park	1
Oman	Masirah	462
Pakistan	Sandspit and Hawkes Bay	1
Total		1114

Appendix B
Satellite tracking information

Table B.1 List of data sources for satellite tracking data used in the determination of distributions of sea turtles within the SWIO region.

Country	Source	Project name	Project URL
France	www.seaturtle.org	Mayotte Isalnd Green Turtles 2005 Islameta Group Dept of Biology University of Pisa	www.seaturtle.org/tracking/index.shtml?project_id=28
France	www.seaturtle.org	Europa Island Green Turtles Islameta Group, Dept of Biology- University of Pisa	www.seaturtle.org/tracking/index.shtml?project_id=13
France	SWOT (2012)		
Kenya	www.seaturtle.org	WWF/KWS Integrated Sea Turtle Conservation Project Kenya	www.seaturtle.org/tracking/?project_id=307
Mozambique	www.seaturtle.org	Maluane/ZSL Turtle Conservation Project in Mozambique: Green Turtles Marine Turtle Research Group	www.seaturtle.org/tracking/index.shtml?project_id=204
Seychelles	www.seaturtle.org	Aldabra Green Turtles Seychelles Island Foundation	www.seaturtle.org/tracking/index.shtml?project_id=712
Seychelles	www.seaturtle.org	Mahe Seychelles Hawksbill Project MCS_Seychelles	www.seaturtle.org/tracking/index.shtml?project_id=277
South Africa	Luschi *et al.* (2006)		
South Africa	De Wet (2012)		
Tanzania	www.seaturtle.org	Post nesting migrations of green turtles nesting in Mafia Island Marine Prak, Tanzania Sea Sense	www.seaturtle.org/tracking/index.shtml?project_id=760

Appendix C
Mitigation measures

Longline fishery

Several types of mitigation measures have been investigated and proposed for the longline industry. These measures include changes to fishing gear and methodology, proper handling and release practices, industry regulations and voluntary communication programs of fleets in the longline fishing operations. Several changes in fishing gear and methodology has been tested with the most effective mitigation measures being changes to hook designs. There are three types of hooks used in the longline industry Japan tuna hooks, circle hooks and J hooks (Beverly *et al.*, 2003). Changes to hook designs aim to reduce the catch rates of sea turtles and to decrease the chances of injury or mortality. The width of the hook will determine whether a sea turtle can swallow the hook. Using larger hooks has the potential to reduce mortality and post-capture mortality in loggerheads due to the fact that these hooks are too large to be swallowed by the sea turtles (Báez *et al.*, 2013, Cambiè *et al.*, 2012, Stokes *et al.*, 2011, Watson *et al.*, 2003). Circle hooks baited with mackerel and sardine bait can reduce the bycatch of loggerheads by as much as 90 % and the bycatch of leatherback by as much as 75 %, compared to J hooks baited with squid (Garrison, 2003, Watson *et al.*, 2004, Watson *et al.*, 2005b). Mackerel bait seemed more efficient at catching swordfish and circle hooks at catching tuna (Watson *et al.*, 2004), mackerel bait however was less efficient for catching tuna than squid (Watson *et al.*, 2004, Watson *et al.*, 2005a). Circle hooks are also less likely to be ingested by sea turtles and can reduce post-capture mortality (Bolten & Bjorndal, 2002, Bolten *et al.*, 2004, Nakano, 2004, Watson *et al.*, 2005b).

The diameter, stiffness, and color of branch lines may also play a role in sea turtle interactions. Larger diameter branch lines were more visible to sea turtles and they were able to avoid them, and stiffer branch lines reduce the possibility of entanglement in the line (Hataway & Mitchell, 2002). Decreasing soak time may also reduce the interaction with sea turtles (Bolten *et al.*, 2004, Watson *et al.*, 2003). Retrieval of lines earlier in the day may reduce capture as captures increased when lines were retrieved later in the day (Bolten & Bjorndal, 2002, Bolten *et al.*, 2004). Eliminating shallow hooks in deep sets also have the potential to decrease bycatch of sea turtles (Beverly *et al.*, 2009, Crowder & Myers, 2001, Gilman *et al.*, 2007, Polovina *et al.*, 2000, Shiode *et al.*, 2005), as sea turtles spend most of their time in the top 100 – 200 m below the surface.

Proper handling and release practices for sea turtles captured in longline fisheries may reduce injury and mortality of sea turtles. Sea turtles captured in longline gear may be alive, dead, or comatose when hauled aboard or alongside vessels during gear retrieval. Additional injury may occur during the hauling process and death or damage can occur from forced submergence, hook penetrating internal organs, or blood vessels or due to subsequent infection caused by the injury. Live sea turtles may also be cut free with hooks still in their throats and mouths with varying lengths of line trailing from the hook. The line can be ingested causing further injury or damage or could result in entanglements or even death by strangulation. Several recommendations have been made for proper handling practices of sea turtles (Balazs *et al.*, 1995, Epperly *et al.*, 2004). Several tools and techniques have been developed to disentangle and de-hook sea turtles that are captured in longline operations in order to increase the chances of post capture survival.

Industry regulations are often used to reduce bycatch of sea turtles. Closure of fisheries may result in the movement of effort to other regions, which may increase bycatch in those regions (James *et al.*, 2005b). Due to trans-boundary movement of sea turtles, management measures should be implemented over a broad geographical range. Seasonal time/area closures can be effective at reducing sea turtle bycatch. However in order for closures of areas to be effective it is imperative to know the distribution of abundance of sea turtles throughout the area and to determine the seasons and areas of higher abundances such as nesting beaches during nesting season and specific migratory routes of sea turtles between nesting and feeding grounds. Fleet communications to reduce sea turtle captures have also proven to be successful at reducing sea turtle captures in fisheries (Gilman *et al.*, 2006b).

Due to the high levels of bycatch of specifically leatherbacks in the region compared to the nesting abundances of the species in the region greater effort should be put into the reduction of leatherback capture in the region. Even though leatherbacks are most frequently externally hooked or entangled in longlines, the use of circle hooks in combination with mackerel bait has proven to be effective in decreasing the bycatch of leatherback turtles (Watson *et al.*, 2005b). Eliminating shallow hooks in sets may reduce interactions even though leatherbacks dive up to 1200 m deep they spend most of their time in the top 200 m below the surface.

Deeper set hooks will thus reduce the possibility of sea turtles being hooked at depth where they most frequently occur. Using larger diameter lines that are more visible to sea turtles and stiffer branch lines may also reduce the possibility on entanglement of sea turtles (Hataway & Mitchell, 2002). It is improbable that all interactions between leatherbacks and longline vessels can be eliminated it is thus imperative that proper handling and release practices are used in order to increase the probability of post-capture survival of leatherbacks. In the SWIO the nesting beaches are already protected by an MPA along the KZN south coast however leatherback frequently move beyond the area closed for fishing vessels. A possible increase in the prohibition area during nesting seasons may decrease the probability of interactions between leatherbacks and longline fishing vessels during the nesting season. Because leatherback feed over wide geographic ranges and show very little preferences for feeding areas it is not possible to propose time/area closures for this species for feeding areas however. Good observer coverage of longline fleets are essential to understand the interaction of sea turtles with the longline fishery in terms of the spatio-temporal variation in catches, that may lead to more accurate estimates of sea turtle interactions with longline operations.

Purse seine fishery

The mortality of sea turtles in the SWIO region is very low compared to other fisheries with estimated 20 sea turtle mortalities in the region. The greatest concern however is the mortalities of sea turtles that occur in association with FADs and ghost fishing as a result of lost or abandoned FADs (Amandé *et al.*, 2008, Clermont *et al.*, 2012) that can have substantial impact on sea turtle populations (Orós *et al.*, 2005, Wilcox *et al.*, 2013, Wilcox *et al.*, 2014). The risk of entanglement with FADs are associated with netting used in construction of these (Delgado de Molina *et al.*, 2007). Alternative and biodegradable designs to FADs are suggested (Franco *et al.*, 2012).

Trawling

Effective mitigation measures have been developed for trawl fisheries, however the implementation and the enforcement of the measures are often problematic (Bourjea *et al.*, 2008, Kiszka & Muir, 2007, Lewison & Crowder, 2007). Considerable reductions in bycatch of sea turtles are possible with the use of TEDs (Brewer *et al.*, 2006, Brewer *et al.*, 1998, Fennessy & Isaksen, 2007). TEDs are gear modifications that reduce the capture of sea turtles in trawl nets (Haas, 2010), with the most common TEDs used in the shrimp fishery being those with inclined metal grids that guide sea turtles to the escape opening (Eayrs, 2007). The use of TEDs in conjunction with the use of Bycatch Reduction Devices (BRDs) has been shown to decrease bycatch in prawn trawl fisheries by as much as 99 %. The use of TEDs and BRDs and effective enforcement of the implementation of these devices will be sufficient to reduce the impact of the trawling industry on sea turtle populations in the SWIO. In the SWIO TEDs are mandatory in most countries, however enforcement of the use of these devices are not effective throughout the region (Bourjea *et al.*, 2008). Proper handling and release practices will also be able to reduce the impact by keeping comatose sea turtles onboard until the sea turtle has sufficiently recovered to be able to swim and surface to breathe (Casale *et al.*, 2004).

Gillnets

Several mitigation measures have been proposed to minimize sea turtle bycatch in gillnet fisheries including visual alerts, time/area closures, smaller net profiles, longer tie downs, float lines without buoys and smaller mesh sizes. Using visual cues to alert non-target species to the presence of gillnets in the water column include changes to the color of the net, illuminating the net, using larger twine sizes, increasing the number of filaments in the nets and using visual markers that are placed along the nets. The use of visual cues have been shown to be effective at reducing the bycatch of sea turtles, however the use of visual cues often also result in a decrease in the catch of target species (Gilman *et al.*, 2010a). Nets illuminated with LED lights, chemical light sticks or UV-lights can reduce green turtle bycatch without reducing the target catches (Wang *et al.*, 2013, Wang *et al.*, 2010). However, the effect of illumination on leatherback catches has not been tested. This is of concern as leatherbacks are thought be attracted to light sticks used in the longline fishery (Witzell, 1999).

Time-area closures have potential to reduce bycatch of sea turtles. This method closes a specific area for a period where bycatch of sea turtles is considered high. However, to implement this method, extensive knowledge of the spatio-temporal patterns of both the bycatch species and the fishery is necessary. This information is often lacking for sea turtle species and artisanal fisheries such as gillnet fisheries.

Nets that are smaller in profile tend to have lower catches of sea turtles, and releasing sea turtles from smaller nets is easier (Gearhart *et al.*, 2009, Price & Salisbury, 2007). Smaller nets may be effective at reducing bycatch of sea turtles due to two reasons. Firstly the net may be stiffer reducing the chances of entanglement and resulting in the sea turtles bouncing off the net rather than being entangled and secondly the smaller nets also reduce the portion of the water column that is being fished an consequently reducing interactions between sea turtles and nets (Price & Salisbury, 2007). Lower profile nets may also reduce mortality in nets by reducing time and effort to disentangle sea turtles from nets, consequently also resulting in less damage to gear (Gearhart & Eckert, 2007). Increasing the length of tie downs or avoiding the use of tie downs have also been shown to decrease entanglement in demersal gillnets (Price & Salisbury, 2007) and higher escape rates for entangled sea turtles (Gearhart & Price, 2003). Using float lines without buoys may also be effective in reducing sea turtles bycatch (Peckham *et al.*, 2009). Larger mesh sizes may also increase sea turtle bycatch (Price & Salisbury, 2007). However, the use of smaller mesh sizes should be carefully implemented as not to increase the catch of small juvenile target species.

Beach seines

Sea turtle bycatch in beach seine nets are often retained for consumption or medicinal or cultural purposes (Kiszka, 2012a, Tietza *et al.*, 2011). This highlights the need to understand and to consider these socio economic and cultural concerns when considering bycatch reduction measures in this fishery.

Appendix D
Productivity and Susceptibility criteria

Productivity criteria

Long-term population trend

Long-term population trend is the population trend over at least one generation (a generation being the age at half of the reproductive lifespan). Sea turtle population dynamics are better represented by long-term trend than recent trend (Chaloupka *et al.*, 2008, NRC, 2010). The most common way of assessing sea turtle population trends is by using the available long term data of annual abundance counts of nesting females (NRC, 2010, Wallace *et al.*, 2011a). The trend in annual numbers of nesting females was thus used as a proxy for long-term population trends.

Recent population trend

The recent population trend of a species is the population trend over the past 5-10 years. Recent trends give a indicates whether there are acute drivers causing population changes, which long-term trend does not show (NRC, 2010, Wallace *et al.*, 2011b). As for long-term trend, recent population trends are based on the trends in the annual numbers of nesting females at rookeries.

RMU size

Annual abundance counts of nesting females are used as a proxy for population size for each of the rookeries. Data on abundance counts of nesting females were obtained from literature and compared to global online databases from State of the World's Sea Turtles – SWOT database (Kot *et al.*, 2013, SWOT, 2006, SWOT, 2007, SWOT, 2008, SWOT, 2009, SWOT, 2010, SWOT, 2011, SWOT, 2012, SWOT, 2013) and IOSEA online reporting facility (www.ioseaturtles.org). Abundance bins for population sizes were generally established by order of magnitude however multiple bins were used to allow for differences in relative abundances of species (Table D.1, Wallace *et al.*, 2011a). Where more than one rookery exists in a RMU, the available abundance values were added together and the RMU was assigned a score based on the cumulative abundances.

Table D.2 Abundance bins reflecting reletive abundances of each species (Wallace *et al.*, 2011a).

Species	Scores				
	Very small (1)	**Small (1.5)**	**Medium (2)**	**Large (2.5)**	**Very large (3)**
Loggerheads	<100	101-1 000	1 001-5 000	5 001-10 000	>10 000
Green turtles	<100	101-1 000	1 001-5 000	5 001-10 000	>10 000
Hawksbills	<10	10-100	101-500	501-1 000	>1 001
Leatherbacks	<10	10-100	101-500	501-1 000	>1 001
Olive ridleys	<100	101-1 000	1 001-10 000	10 001-100 000	>100 000

Genetic diversity

Genetic diversity is the number of known or inferred genetic stocks within an RMU. The genetic diversity is based on the genetic distinctiveness of rookeries that is in turn based on mitochondrial DNA analysis. The genetic diversity of an RMU indicates the genetic uniqueness of an RMU and is an indication of the risk of losing a distinct isolated genetic stock. Extirpation of populations would result in the permanent loss of genetic variability within a species, which could reduce its resilience and resistance to extinction (Carlton *et al.*, 1999, Gilman *et al.*, 2011).

Age at maturity

Age at maturity is the age at first reproduction of a species. Age at maturity is indicative of the longevity of a species and tends to be positively correlated with maximum age and natural mortality (Patrick *et al.*, 2010). The higher the age at maturity the lower the productivity of a species tends to be. Where values for this criterion were not available for populations in the SWIO, it was inferred from studies outside the region, assuming that it will be very similar across populations of the same species. The age at maturity for breeding females are used, as no data are available for age at maturity for male sea turtles, it is however assumed that this parameter is not gender specific.

Maximum age

Maximum age is an indication of the longevity of individuals and an indication of the natural mortality rate, with low levels of natural mortality being negatively correlated with high maximum ages (Patrick *et al.*, 2010). Long-lived species have longer reproductive spans (Jennings & Beverton, 1991), and tend to have low maximum population growth rates making them more vulnerable to exploitation due to the lower recovery potential (Hutchings

2001a, Hutchings and Reynolds). Maximum age was inferred from the maximum reproductive age. Thus maximum age (A_{max}) is calculated as follows:

$$\boldsymbol{A_{max} = A_m + R_L} \qquad \textbf{Equation D.1}$$

where A_m is the age to maturity and R_L is the maximum reproductive lifespan. Very few data are available for the maximum reproductive lifespan of sea turtles in the SWIO and these were inferred from populations outside the region assuming that it will be similar for different populations of the same species.

Generation length

Generation length (G_L) is the age at half of the maximum reproductive life span (Pianka 1974). Calculated as follows:

$$\boldsymbol{G_L = A_m + \frac{1}{2} R_L} \qquad \textbf{Equation D.2}$$

where A_m is the age to maturity and R_L is the reproductive lifespan. This is an important criterion for productivity as it reflects the longevity of individuals (as for maximum age) as well as the turnover rate of individuals in the breeding population.

Natural survivorship: Nest success

Nest success is the percentage of nests in a rookery that successfully produces hatchlings. The number of nests produces by a population may be high but the reproductive output is decreased when a high proportion of these nest do not successfully produce hatchlings. Nest success is influenced by local environmental conditions and factors such as poaching of eggs, predation by natural or introduced predators and, beach erosion, and inundation of nests.

Natural survivorship: Hatching and emergence success

Hatching and emergence success is the percentage of eggs per nest that emerge as hatchlings. A female may lay a significant number of eggs within a breeding season, however the resultant reproductive capacity is influence by the percentage of these eggs that successfully develop to produce hatchlings. This criterion is also influenced by local conditions, as is the case with nest success.

Number of eggs per nest

The average number of eggs a female lays per nest gives an indication of the potential reproductive capacity of the female, with greater the number of eggs per nest, the higher the potential reproductive capacity of a female. Where data for specific rookeries or RMUs are not available, this criterion was inferred from species-specific data from published literature. It is assumed that this will be similar to other populations within the same species.

Number of clutches per female per season

The number of clutches a females lays per nesting season influences her overall reproductive output, with the greater the number of clutches laid, the higher the potential reproductive capacity of a female. Where data for specific rookeries were not available, data were inferred from species-specific data or form adjacent rookeries. It is assumed that this characteristic will be similar to other populations of the same species.

Remigration interval

The remigration interval is the number of years between successive nesting seasons for a female. A female with a shorter remigration interval has a higher potential reproductive output compared to a female with a longer migration interval. Remigration intervals are often dependent upon food availability, environmental factors, and migration distance between foraging grounds and nesting beaches (Bjorndal, 1985, Solow *et al.*, 2002, Troeng & Chaloupka, 2007). Remigration interval does offer important insight into the productivity of the population and population density relative to carrying capacity (Saba *et al.*, 2007, Troeng & Chaloupka, 2007). Breeding rates of male sea turtles however have been poorly studied and it is believed that males may breed at greater frequency than females and that a substantial portion of males may breed annually (Hamann *et al.*, 2003). Remigration intervals were thus based on female remigration intervals.

Susceptibility criteria

Horizontal overlap between species distribution and fishery

The horizontal overlap pertains to the extent of horizontal overlap between the fishery and the known distribution of a species. Some degree of horizontal overlap is necessary for a fishery to influence a stock. This attribute is based on the assumption that the greater the horizontal overlap between a fishery and the distribution of a species the greater the possibility that an individual of the species may interact with a fishery. This criterion is indicative of the possible encounter rate, but overlap cannot be used in isolation of some form of bycatch calculation. Several methods exist for calculating the overlap with fisheries however, in the present study due to the lack of data on sea turtle densities in feeding grounds and insufficient information about migratory pathways a simple overlap calculation was computed (See Chapter 3 and 4). Horizontal overlap between fishery distribution and sea turtle distributions was calculated as the percentage of sea turtle distribution that falls within the area of fishing effort.

Vertical overlap between species distribution and fishery

This attribute pertains to the overlap between the positions of the species within the water column relative to that of the position of the fishing gear within the water column. The diving depth of sea turtles varies between feeding and nesting sites as well as along migration routes (See Chapter 2 for species-specific details). The diving depth data were compared with the depth of fishing gear in order to determine the degree of overlap (see chapter 3 and 4). Overlap with maximum diving depth of species was used for comparison.

Confidence estimate in distribution data

The confidence in distribution data of a species is based upon the number of individuals of the species that have been tracked by satellite transmitters in the region. Higher confidence levels were assigned to distribution data where more individuals were tracked using satellite transmitters.

Geographic concentration

The geographic concentration is the extent to which a species is concentrated in small areas. The rationale behind including this parameter is with relatively even distributions across its range may be less susceptible to fishing than a stock that is highly concentrated in specific areas. Current knowledge on sea turtle densities in both feeding and nesting grounds are sparse. It was thus not possible to score this criterion.

Bycatch mortality relative to natural mortality

Sea turtle bycatch data are often sparse, heterogeneous in quality and biased to certain geographic locations. The methods used to calculate bycatch estimates for each of the fisheries are given in relevant chapters: Chapter 3 for longline and purse seine fisheries, Chapter 4 for prawn trawl, gillnet and beach seine fisheries. One of the greatest drawbacks of late sexual maturity in species such as sea turtles, is that it increases the probability of dying before reproducing (Heppell *et al.*, 2003b). Survivorship varies among age classes, species and populations (Table B.2), with the level of survivorship being highly dependent upon the anthropogenic impacts that a rookery is facing (Heppell *et al.*, 2003b). The annual survivorship of more pristine rookeries that experience low levels of human impact are thus more likely to reflect natural survivorship values (Heppell *et al.*, 2003b). The natural survivorship from eggs to hatchlings is often high (Anastácio *et al.*, 2014, De Wet, 2012, Innocenzi *et al.*, 2010), however this varies from year to year and is dependent upon factors such as predation and environmental conditions at nesting beaches. The survival of post-hatchlings in their first year is however very low, whereas the natural survivorship of juveniles and sub-adults is higher, yet not as high as that for adults (Heppell *et al.*, 2003b).

It has been shown that the long term growth rate of sea turtles are more sensitive to a 10 % change in the survival of large juveniles than it is to a 10 % change in the hatchling or adult survival rate, because most of the population will consist of large juveniles when the population is at a stable age distribution (Heppell *et al.*, 2000). Increases in hatchling and egg survival cannot compensate for increased mortality of sub adults and adults, however input from the early stages are important to maintain recruitment to older stages. There is large data gaps regarding the survivorship data of populations in the SWIO and were inferred from survivorship data published for other regions. It has been established that larger sea turtles have lower natural mortality (5-10%). Generic mortality values were thus applied.

Assuming a sex ratio of 1:1 males to females and that adults constitute 10 % of the normal populations of sea turtles, 30 % of the annual number of nesting females probably represents 1.5-3 % of adult females. (Heppell, 2012). Threat posed by each of the gear types was compared to natural mortality (assuming that natural mortality of species in the SWIO is comparable to those in other ocean regions).

Table B.3 Survivorship data for different age classes (adult, sub-adult, and juveniles) of four species of sea turtles from various populations outside of the SWIO.

Species	Area	Adult	Sub-adult	Juvenile	References
Green turtle	Australia – Great Barrier Reef	0.95	0.85	0.88	Chaloupka and Limpus (2005)
Green turtle	Gulf of California	0.97		0.58	Seminoff *et al.* (2003)
Green turtle	Bahamas			0.76-0.89	Bjorndal *et al.* (2003b)
Green turtle	Nicaragua	0.55		0.55	Campbell and Lagueux (2005)
Green turtle	Tortuguero	0.82 0.85			Campbell and Lagueux (2005) Troeng and Chaloupka (2007)
Hawksbill	Australia - Western	0.95			Prince and Chaloupka (2012)
Hawksbill	Long Island (USA)	0.95			Kendall and Bjorkland (2001)
Hawksbill	Australia – Great Barrier Reef	0.92$^{\#}$ 0.72*		0.93$^{\#}$ 0.78*	Bell *et al.* (2012)
Leatherback	Costa Rica	0.65			Spotila *et al.* (2000)
Leatherback	French Guiana	0.91 0.85			Rivalan *et al.* (2005) Pilcher and Chaloupka (2013)
Leatherback	US Virgin Islands	0.93		0.63	Dutton *et al.* (2005)
Leatherback	Florida	0.96			Stewart (2007)
Leatherback	St Croix		0.004 – 0.02$^{\$}$		Eguchi *et al.* (2006a)
Loggerhead	Little Cumberland Island - USA	0.81		0.68	Frazer (1983) Frazer (1987)
Loggerhead	Australia - Queensland	0.91 0.78	0.83	0.89	Heppell *et al.* (1996a)
Loggerhead	USA – western north Atlantic			0.89	NMFS (2001b)
Loggerhead	Australia Great Barrier Reef	0.88		0.92	Chaloupka and Limpus (2002)
Loggerhead	Mediterranean		0.73		Casale *et al.* (2007)
Loggerhead	USA – western north Atlantic	0.74-0.87$^{\&}$	0.75-0.88$^{\%}$ 0.68-0.7$^{\%}$		Heppell *et al.* (2003a)
Loggerhead	Azores			0.91	Bjorndal *et al.* (2003a)

$^{\#}$Females *Males $^{\$}$All age classes $^{\%}$Juveniles and sub-adults $^{\&}$Sub-adults and adults

Post-capture mortality

Post-capture mortality relates to the survival rate of captured sea turtles after being released alive and does not include mortality that occurs during the capture. Post capture mortality is due to injury suffered during capture or a decrease in physical condition that results in mortality after release. Reliable estimates of post-capture mortality are not available and this criterion was not scored in the analysis.

Seasonal changes in overlap

Seasonal migrations to or from a fishery could affect the overlap between the stock and the fishery. This also relates to areas where the location of the fishery will change between seasons as this might increase or decrease overlap with bycatch species. Due to the migratory nature of sea turtles, there is a possibility of a variation in the seasonal overlap of sea turtles with fishing operations. Sea turtles are often present at nesting beaches only during the nesting season and migrate back to feeding grounds after the nesting season. During nesting seasons, sea turtles will be distributed adjacent to nesting beaches increasing risk of capture in inshore-fisheries. Outside of the nesting season, the risk of capture maybe higher at feeding grounds due to the migration of sea turtles back from nesting grounds to feeding grounds. Higher risk across migratory pathways will occur during migrations between nesting and feeding grounds. These seasonal changes in overlap with fisheries activities were not possible to score due to the paucity of data.

Size class most frequently caught in fishing operation

Sea turtle populations are more vulnerable to impacts that result in the mortality of older age classes as the relative reproductive value of sea turtles change radically from hatchlings to adults (Crouse *et al.*, 1987, Heppell, 1998). However, the different age classes also show different susceptibility rates to being caught in fisheries. The age classes most commonly caught by fisheries vary between fisheries and even within fisheries. The age class most predominantly taken by a fishery can be dependent on the gear used and the area fished (Wallace *et al.*, 2010b). When attempting to define mortality limits for fisheries the age classes caught are important as reducing the amount of adults killed will have a greater positive effect on population recovery than reducing the amount of hatchlings killed (Crouse *et al.*, 1987, Heppell *et al.*, 2005). This criterion is however not scored due to paucity of data.

Appendix E

Sensitivity analysis of the PSA

To assess the sensitivity, a hypothetical stock was tested under several different scenarios. In the first scenario, the effect of changing attributes scores was tested with a varying number of attributes. This was investigated for both changes in the number of P and S attributes. When susceptibility scores were changed the productivity score was held constant at medium (2) and vice versa. In the second scenario, all P scores were held constant at medium (2) and S scores manipulated. Vulnerability scores were calculated for the successive increases in the number of S scores that were changed. Three different changes in S scores were investigated; (i) changing S scores from 1 to 2, (ii) changing scores from 1 to 3 and, (iii) changing scores from 2 to 3. In the third scenario, all S scores were held constant at 2 and P scores were changed. Vulnerability scores were calculated for successive increases in the number of P scores that were changed and three different changes were investigated as per the second scenario method. In the fourth scenario, the weightings of one of the attributes were changed and the rest held constant. Weightings were changed to 2x, 3x and 4x the weighting of the other attributes. Changes in the weighted attributes were investigated with respect to each other.

Results

Changes in vulnerability scores are greater in analyses where fewer attributes are used but the same number of attributes is changed. This same applies to both changes in susceptibility (Fig. E.1) and productivity (Fig. E.2).

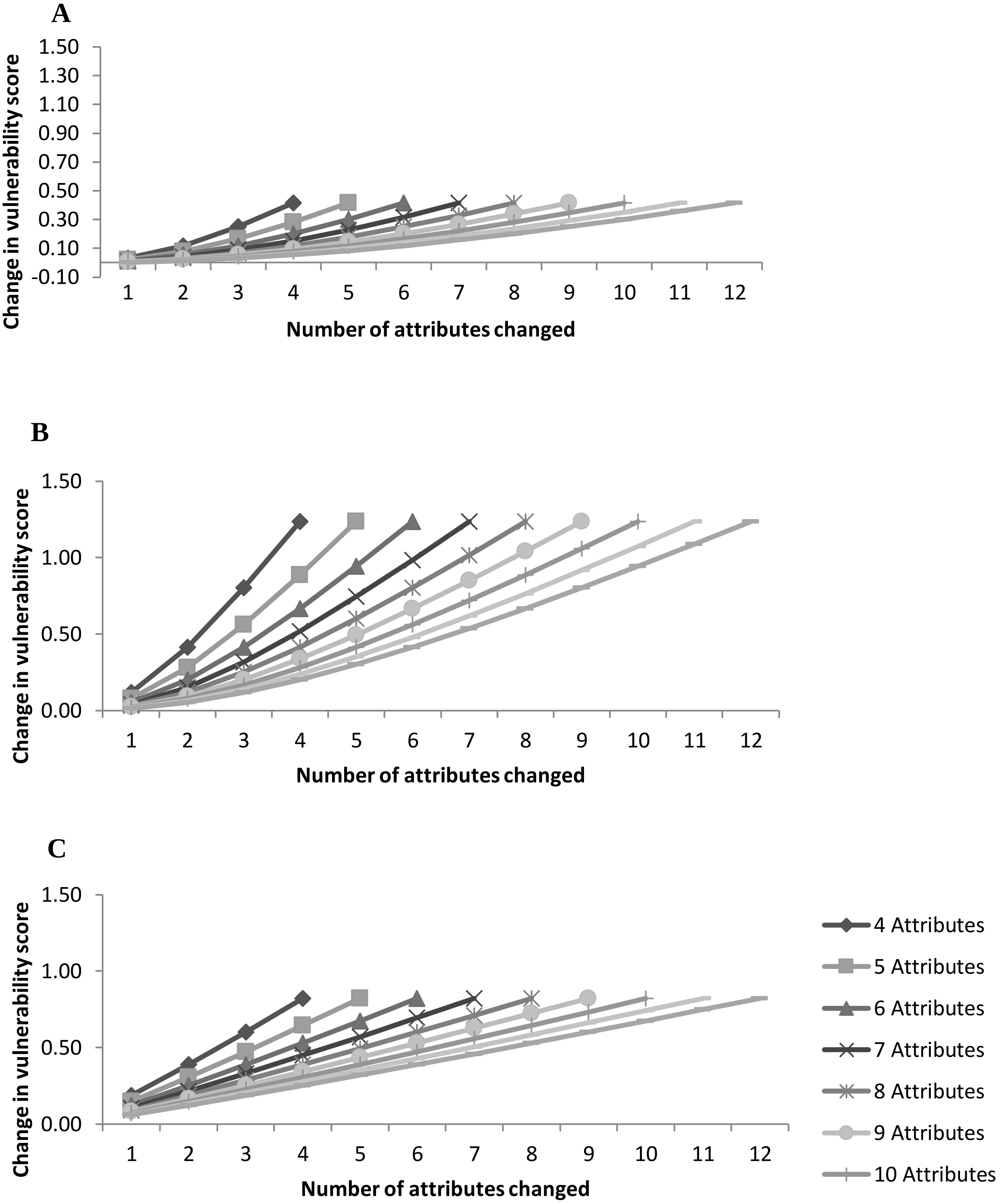


Figure E.1 Change in vulnerability scores when susceptibility scores are change from A) 1 to 2, B) 1 to 3, C) 2 to 3 for productivity-susceptibility analyses with different numbers of changed attributes and different number of total attributes used in the analyses.

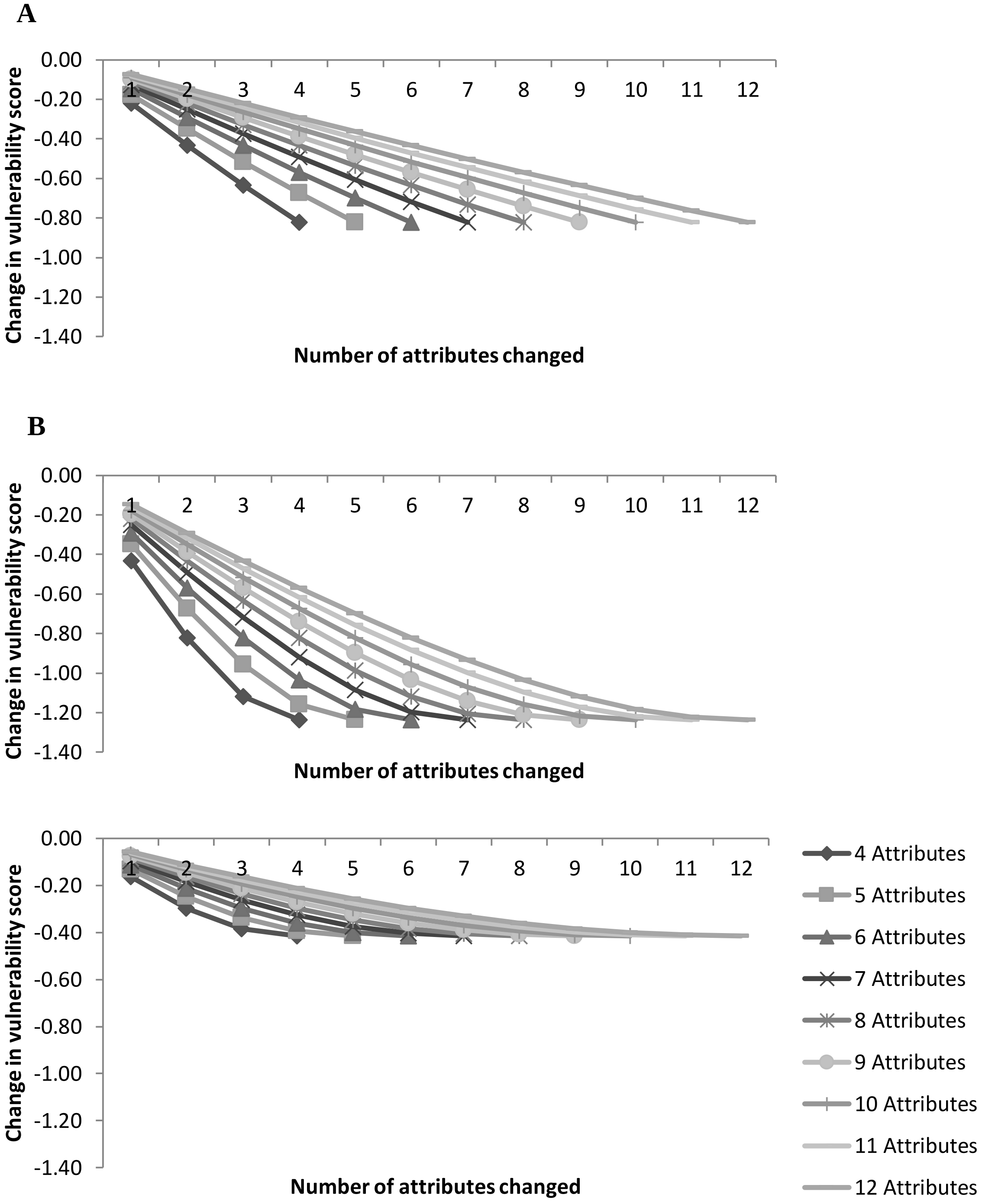


Figure E.2 Change in vulnerability scores when productivity scores are change from A) 1 to 2, B) 1 to 3, C) 2 to 3 for productivity-susceptibility analyses with different numbers of changed attributes and different number of total attributes used in the analyses.

Vulnerability scores have larger changes when more attributes are changed than when fewer attributes are changed (Fig E.3). Changes in vulnerability scores are greater for PSAs that have higher initial susceptibility scores when the same number of susceptibility scores is changed and the number of attributes used in the analysis is kept constant (Fig. E.3). A PSA with an initial attribute score of 2 had a greater change in vulnerability when the score was changed to 3 then where the initial susceptibility score was 1 and changed to 2.

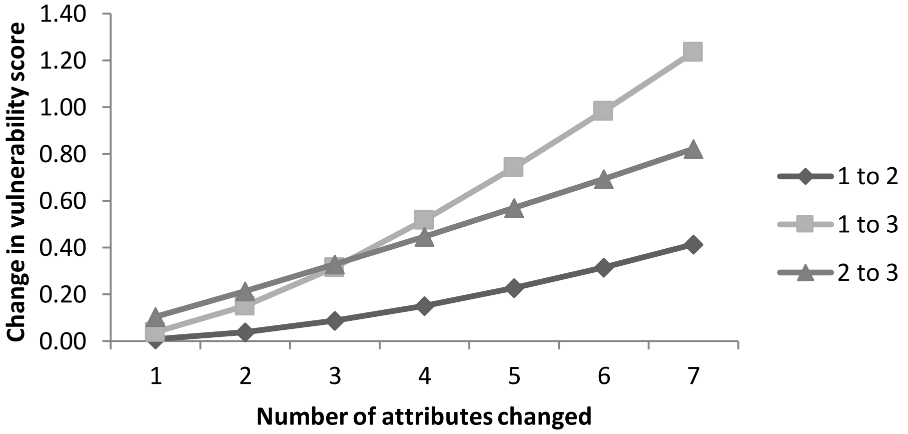


Figure E.3 Change in vulnerability score when susceptibility scores are changed (from 1 to 2, 1 to 3, and 2 to 3), when the number of attributes are held constant.

Vulnerability scores decrease when the productivity scores are increased (Fig. E.4). The decrease in vulnerability scores is greatest when changes in productivity scores are greater. However when initial productivity scores were higher a smaller change in vulnerability is seen than when initial productivity scores are lower. When productivity scores are changed from 2 to 3 there is a smaller change in vulnerability than when productivity scores are changed from 1 to 2 even though the increment in productivity scores is the same.

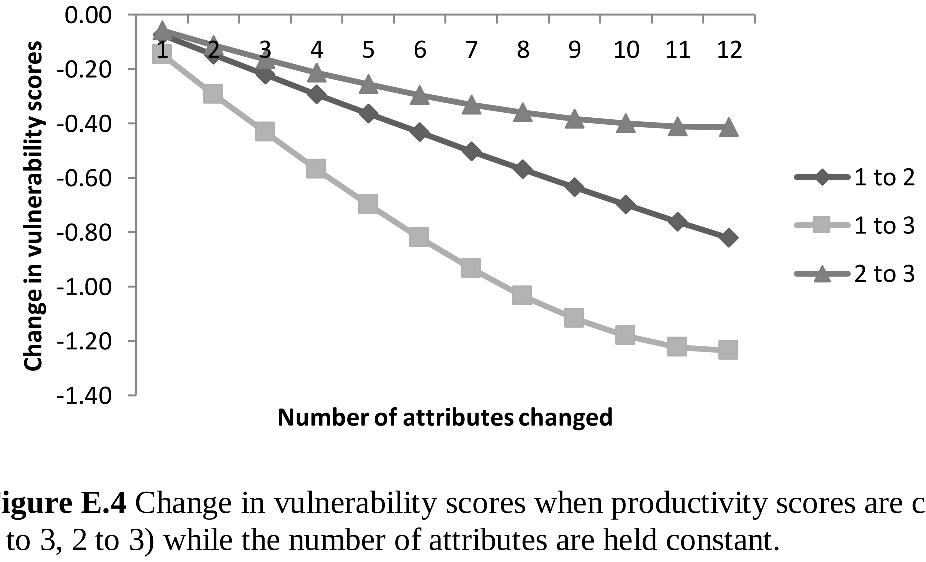


Figure E.4 Change in vulnerability scores when productivity scores are changes (from 1 to 2, 1 to 3, 2 to 3) while the number of attributes are held constant.

In the fourth scenario, it is evident that when changing attributes with higher weighting that the change in vulnerability scores are substantially higher than when changing attributes with lower weightings. However higher weighted criteria will also result in smaller changes in vulnerability scores when the lower weighted criteria are changed. The higher weighting of the other criteria will thus overshadow changes in lower weighted criteria. When higher weighted criteria's scores are kept unchanged and lower weighted criteria scores are changed, the change in vulnerability score is smaller the higher the weighting of the higher weighted criteria (Fig. E.5 and E.6).

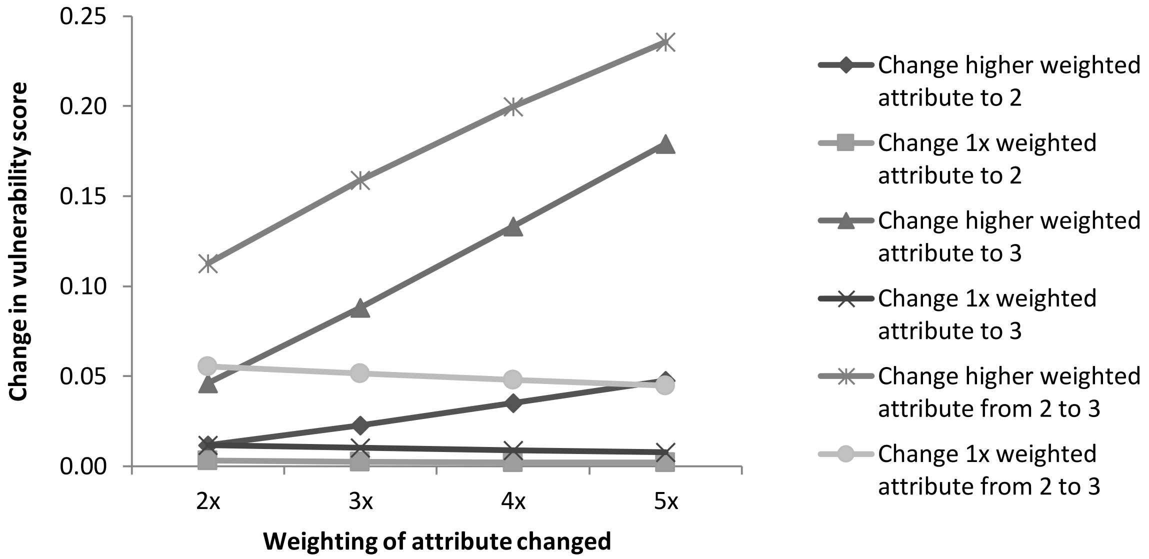


Figure E.5 Change in vulnerability scores when susceptibility attributes of different weightings are changed (from 1 to 2, 1 to 3, 2 to 3).

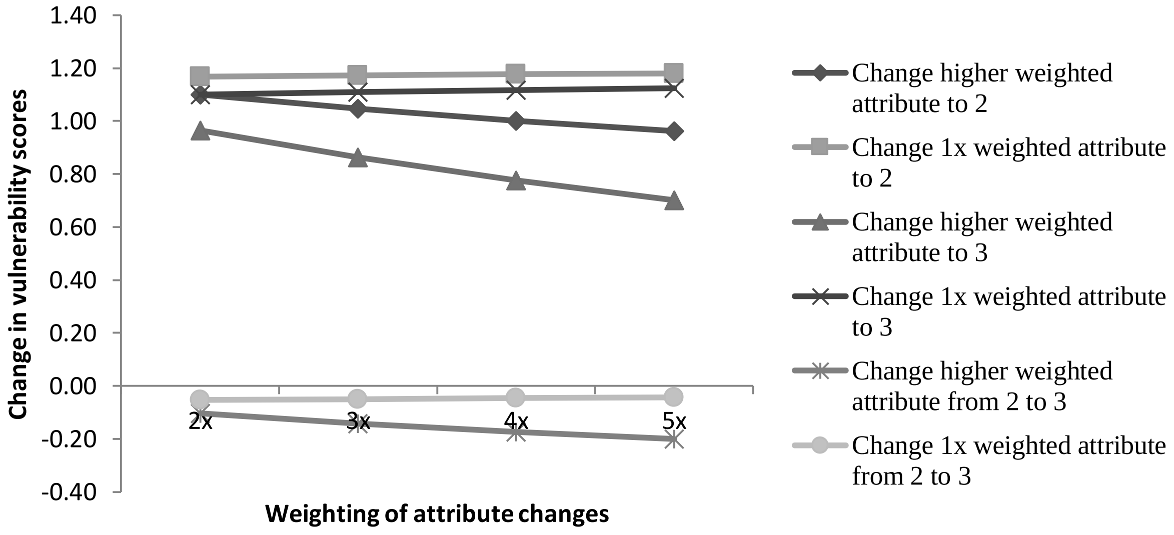


Figure E.6 Change in vulnerability scores when productivity attributes of different weightings are changed (from 1 to 2, 1 to 3, 2 to 3).

Sensitivity analysis revealed that PSAs using fewer attributes in the calculations are more sensitive to changes in the attribute scores. This is evident from the higher relative changes in overall vulnerability scores where fewer attributes are used. It is thus essential to make PSAs as complete as possible, including attributes with lower quality data if necessary. The inclusion of lower quality data however should only be considered where the number of attributes used in an assessment is very few. Similar results for all tests were found by (Ormseth & Spencer, 2011)

A species with a lower initial susceptibility score will have slower changes in the vulnerability score than a species with higher initial susceptibility. When the number of attributes is held constant and susceptibility scores are changed, there is a larger change in vulnerability when the initial score was higher compared to a lower initial score when the increment in attribute scores is similar. If a PSA is intended for periodical review, it should be kept in mind that large changes in susceptibility may lead to smaller changes in vulnerability of a stock when the initial susceptibility is low (Ormseth & Spencer, 2011).

A species with a higher initial productivity score will have slower changes in vulnerability than a species with a lower initial productivity score. When the number of attributes is held constant and productivity scores are changed there is a larger change in vulnerability when the initial productivity score was lower compared to a higher initial score when increments in the attribute scores are similar. When PSAs are used for periodic review, it should be kept in mind that small changes in productivity for species with higher initial productivity may lead to small changes in the vulnerability score.

PSAs can be seen as largely insensitive to changes in attribute scores and large changes in attribute scores are needed for large changes in the vulnerability of a species. This is evident when the number of total attributes in an assessment is held constant and productivity or susceptibility scores are changed, a larger change (for example from 1 to 3 compared to a change from 1 to 2) is necessary for a large change in the vulnerability scores. When continuously monitoring a specific species and factors the process might not capture important changes in vulnerability. In order to overcome this, those attributes that are seen as more important in the assessment of either productivity or susceptibility were given higher weightings than others. Thus, changes in those attributes will result in greater changes in vulnerability than changes in attributes with lower weighting. When attributes with lower weightings are changed, these changes are overshadowed by the attributes with higher weightings. Care should thus be exercise when weighting criteria in order to apply weightings to the most important criteria.

Appendix F
Data quality scores

Productivity attributes

Table F.1 Data quality scores for each of the productivity attributes of each of the species of sea turtles, including the weighted average score and weighted standard deviation.

Characteristic	Weight	Green turtles	Hawksbills	Leatherbacks	Loggerheads	Olive ridleys
Recent population trend	2	1	1	1	1	1
Long term population trend	2	1	1	1	1	1
RMU size/ Population size	3	1	1	1	1	1
Genetic diversity	1	1	5	1	1	5
Age at maturity	1	4	4	4	1	4
Natural mortality : Nest success	1	1	1	1	1	5
Natural mortality: Emergence success	1	1	1	1	1	5
Number of eggs per female per clutch	1	1	1	1	1	3
Number of clutches per female per season	1	1	1	1	1	5
Remigration Interval	1	1	1	1	1	5
Reproductive lifespan	1	4	4	1	1	5
Generation length	1	4	4	3	1	5
Weighted average data quality score (Q)		1.56	1.81	1.31	1.00	3.06
Weighted standard deviation (SD_W^2)		1.22	1.49	0.88	0.00	1.97

Susceptibility attributes

Table F.2 Data quality scores for susceptibility attributes used in the assessment of the longline industry giving the score per attribute per species and showing the weighted average and standard deviation of the scores.

Characteristic	Weight	Green turtles	Hawksbills	Leatherbacks	Loggerheads	Olive ridleys
Horizontal overlap	1	1	1	1	1	1
Vertical overlap	1	3	3	3	3	3
Confidence estimate	1	1	1	1	1	1
Bycatch estimate	3	2	2	2	2	2
Weighted average data quality score (Q)		1.83	1.83	1.83	1.83	1.83
Weighted standard deviation (SD_W^2)		0.79	0.79	0.79	0.79	0.79

Table F.3 Data quality scores for susceptibility attributes used in the assessment of the purse seine industry in Scenario 2 giving the score per attribute per species and showing the weighted average and standard deviation of the scores.

Characteristic	Weight	Green turtles	Hawksbills	Leatherbacks	Loggerheads	Olive ridleys
Horizontal overlap	1	1	1	1	1	1
Vertical overlap	1	3	3	3	3	3
Confidence estimate	1	1	1	1	1	1
Bycatch estimate	3	2	2	2	2	2
Weighted average data quality score (Q)		1.83	1.83	1.83	1.83	1.83
Weighted standard deviation (SD_W^2)		0.79	0.79	0.79	0.79	0.79

Table F.4 Data quality scores for susceptibility attributes used in the assessment of the prawn trawl industry in Scenario 2 giving the score per attribute per species and showing the weighted average and standard deviation of the scores.

Characteristic	Weight	Green turtles	Hawksbills	Leatherbacks	Loggerheads	Olive ridleys
Horizontal overlap	1	1	1	1	1	1
Vertical overlap	1	3	3	3	3	3
Confidence estimate	1	1	1	1	1	1
Bycatch estimate	3	4	4	4	4	4
Weighted average data quality score (Q)		2.83	2.83	2.83	2.83	2.83
Weighted standard deviation (SD_W^2)		1.55	1.55	1.55	1.55	1.55

Table F.5 Data quality scores for susceptibility attributes used in the assessment of the gillnet industry in Scenario 2 giving the score per attribute per species and showing the weighted average and standard deviation of the scores.

Characteristic	**Weight**	**Green turtles**	**Hawksbills**	**Leatherbacks**	**Loggerheads**	**Olive ridleys**
Horizontal overlap	1	3	3	3	3	3
Vertical overlap	1	3	3	3	3	3
Confidence estimate	1	1	1	1	1	1
Bycatch estimate	3	2	2	2	2	2
Weighted average data quality score (Q)		2.17	2.17	2.17	2.17	2.17
Weighted standard deviation (SD_W^2)		0.79	0.79	0.79	0.79	0.79

Table F.6 Data quality scores for susceptibility attributes used in the assessment of the beach seine industry in Scenario 2 giving the score per attribute per species and showing the weighted average and standard deviation of the scores.

Characteristic	**Weight**	**Green turtles**	**Hawksbills**	**Leatherbacks**	**Loggerheads**	**Olive ridleys**
Horizontal overlap	1	3	3	3	3	3
Vertical overlap	1	3	3	3	3	3
Confidence estimate	1	1	1	1	1	1
Bycatch estimate	3	2	2	2	2	2
Weighted average data quality score (Q)		2.17	2.17	2.17	2.17	2.17
Weighted standard deviation (SD_W^2)		0.79	0.79	0.79	0.79	0.79

Appendix G
Productivity and susceptibility scores

Productivity

Table G.4 Summary of scores assigned for each productivity attribute for all species. Overall weighted average scores are given with the associated standard deviation (SD).

Productivity criteria	Weight	Green turtles	Hawksbills	Leatherbacks	Loggerheads	Olive ridleys
Long term population trend	2	1	1	2	3	1
Recent population trend	2	3	1	2	3	1
RMU size/ Population size	3	3	3	1.5	1.5	2
Genetic diversity	1	3	1	1	1	1
Age at maturity	1	1	1	3	1	3
Reproductive lifespan	1	2	3	1	2	1
Generation length	1	2	2	3	1	1
Remigration Interval	1	2	3	2	3	1
Number of clutches per female per season	1	1	2	3	1	1
Number of eggs per female per clutch	1	3	3	2	2	2
Natural mortality : Nest success	1	3	3	3	3	1
Natural mortality: Emergence success	1	3	3	2	2	1
Weighted productivity score (P)		2.50	2.29	2.04	1.89	1.43
Weighted standard deviation (SD_W^2)		0.76	0.92	0.72	0.82	0.65

Susceptibility

Table G.1 Data used in assigning susceptibility scores for each characteristic together with the associated susceptibility score (S) for the longline fishery. Weighted average susceptibility scores are given per species as well as the standard deviation.

Susceptibility criteria	**Weight**	**Green turtles**		**Hawksbills**		**Leatherbacks**		**Loggerheads**		**Olive ridleys**	
		Data	**S**	**Data**	**S**	**Data**	**S**	**Data**	**S**	**Data**	**S**
Horizontal overlap	1	100%	3	100%	3	99%	3	100%	3	93%	3
Vertical overlap	1	100%	3	100%	3	31%	2	100%	3	98%	3
Confidence estimate	1	120	1	12	2	26	2	22	2	1	3
Bycatch mortality	3	42	1	68	1	57	2	68	1	0.00	1
Weighted susceptibility score		1.67		1.83		2.17		1.83		2.00	
Weighted standard deviation (SD_W^2)		1.09		1.04		0.43		1.04		1.15	

Table G.2 Data used in assigning susceptibility scores for each characteristic together with the associated susceptibility score (S) for the purse seine fishery. Weighted average susceptibility scores are given per species as well as the standard deviation.

Susceptibility criteria	**Weight**	**Loggerheads**		**Green turtles**		**Leatherbacks**		**Hawksbills**		**Olive ridleys**	
		Data	**S**	**Data**	**S**	**Data**	**S**	**Data**	**S**	**Data**	**S**
Horizontal overlap	1	54%	2	38%	2	30%	2	43%	2	68%	3
Vertical overlap	1	100%	3	100%	3	23%	1	100%	3	74%	3
Confidence estimate	1	22	2	120	1	26	2	12	2	1	3
Bycatch mortality	3	34	1	49	1	2	1	62	1	112	1
Weighted susceptibility score		1.67		1.50		1.33		1.67		2.00	
Weighted standard deviation (SD_W^2)		0.86		0.88		0.54		0.86		1.15	

Table G.3 Data used in assigning susceptibility scores for each characteristic together with the associated susceptibility score (S) for the prawn trawl fishery. Weighted average susceptibility scores are given per species as well as the standard deviation.

Susceptibility criteria	Weight	Green turtles		Hawksbills		Leatherbacks		Loggerheads		Olive ridleys	
		Data	Score	Data	Score	Data	Score	Data	Score	Data	Score
Horizontal overlap	1%	0.45%	1	0.41%	1	0.63%	1	0.50%	1	0.47%	1
Vertical overlap	1%	100.00%	3	100.00%	3	100.00%	3	100%	3	100.00%	3
Confidence estimate	1%	120	1	12	2	26	2	22	2	1	3
Bycatch mortality	3%	1677	1	419	1	3	1	419	2	112	1
Weighted susceptibility score		1.33		1.50		1.50		2.00		1.67	
Weighted standard deviation (SD_W^2)		0.86		0.88		0.88		0.67		1.09	

Table G.4 Data used in assigning susceptibility scores for each characteristic together with the associated susceptibility score (S) for the gillnet fishery. Weighted average susceptibility scores are given per species as well as the standard deviation.

Susceptibility criteria	Weight	Green turtles		Hawksbills		Leatherbacks		Loggerheads		Olive ridleys	
		Data	Score	Data	Score	Data	Score	Data	Score	Data	Score
Horizontal overlap	1%	1.38%	1	1.29%	1	1.63%	1	1.55%	1	1.39%	1
Vertical overlap	1%	100.00%	3	100.00%	3	100.00%	3	100.00%	3	100.00%	3
Confidence estimate	1%	120	1	12	2	26	2	22	2	1	3
Bycatch mortality	3%	30887	3	2520	2	86	3	5248	3	1366	3
Weighted susceptibility score		2.33		2.00		2.50		2.50		2.67	
Weighted standard deviation (SD_W^2)		1.09		0.67		0.88		0.88		0.86	

Table G.5 Data used in assigning susceptibility scores for each characteristic together with the associated susceptibility score (S) for the beach seine fishery. Weighted average susceptibility scores are given per species as well as the standard deviation.

Susceptibility criteria	**Weight**	**CM**		**EI**		**DC**		**CC**		**LO**	
		Data	**Score**	**Data**	**Score**	**Data**	**Score**	**Data**	**Score**	**Data**	**Score**
Horizontal overlap	1%	0.28%	1	0.26%	1	0.40%	1	0.32%	1	0.27%	1
Vertical overlap	1%	100.00%	3	100.00%	3	100.00%	3	100.00%	3	100.00%	3
Confidence estimate	1%	120	1	12	2	26	2	22	2	1	3
Bycatch mortality	3%	4784	2	1181	2	0	1	2901	3	904	2
Weighted susceptibility score		1.83		2.00		1.50		2.50		2.17	
Weighted standard deviation (SD_W^2)		0.79		0.67		0.88		0.88		0.79	

Appendix H
PSA graphs per fishery

PSA graphs where alternative threat scores were included in the susceptibility analysis

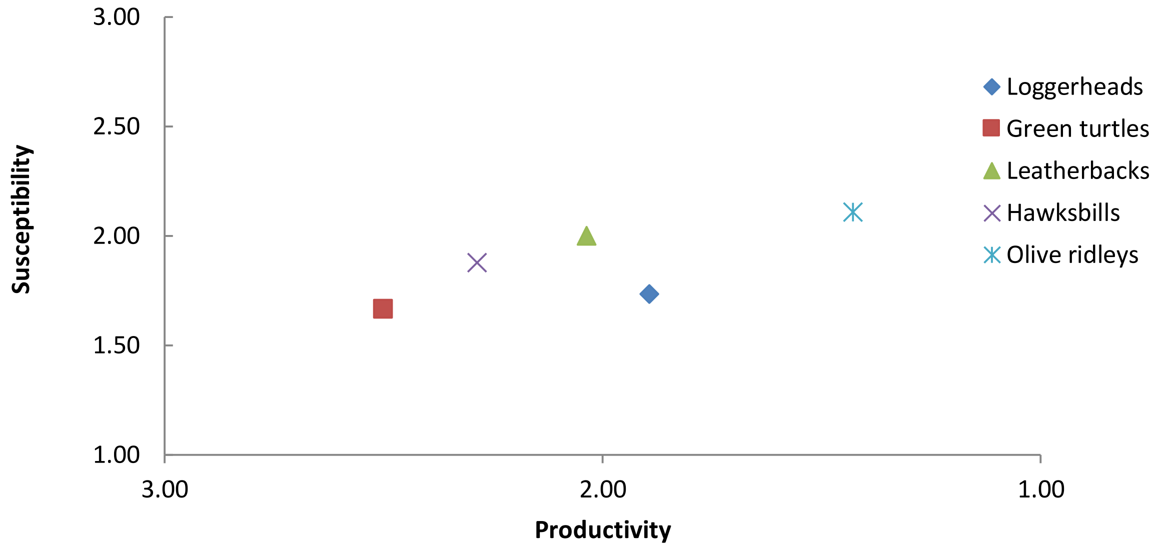


Figure H.1 Vulnerability graph of the longline fishery using susceptibility scores where the alternative threat score was included in the susceptibility analysis.

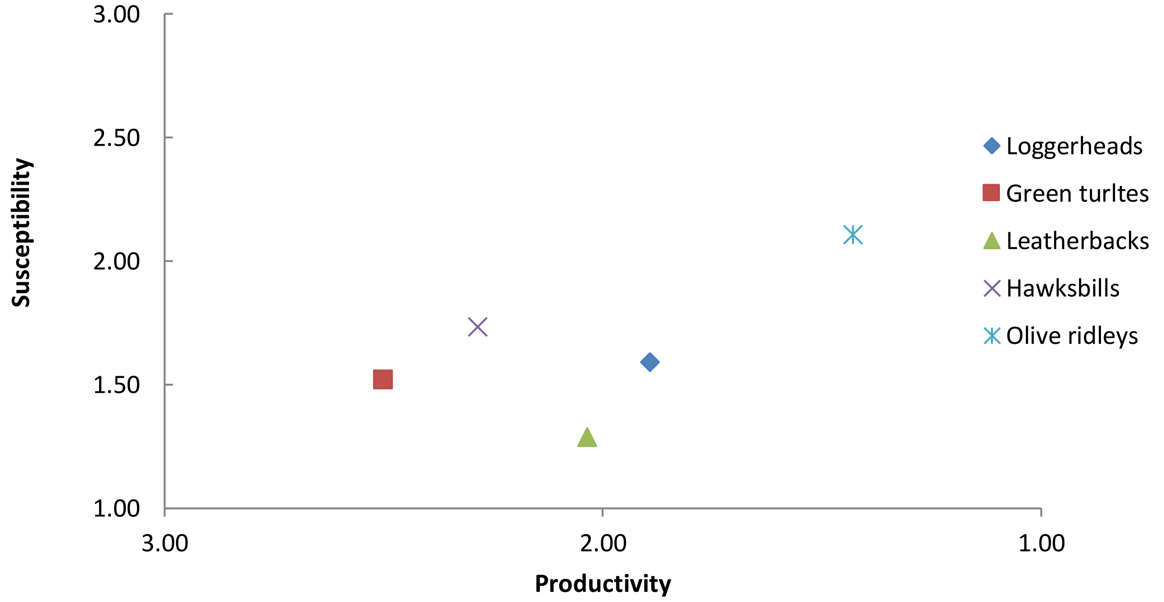


Figure H.2 Vulnerability graph of the purse seine fishery using susceptibility scores where the alternative threat score was included in the susceptibility analysis.

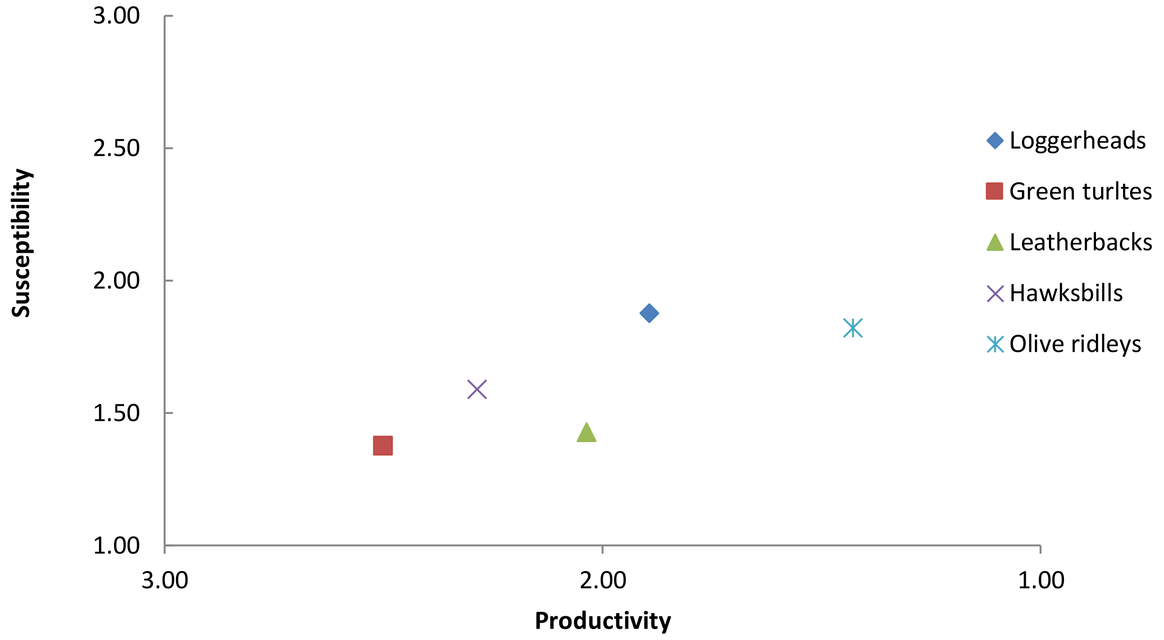


Figure H.3 Vulnerability graph of the prawn trawl fishery using susceptibility scores where the alternative threat score was included in the susceptibility analysis.

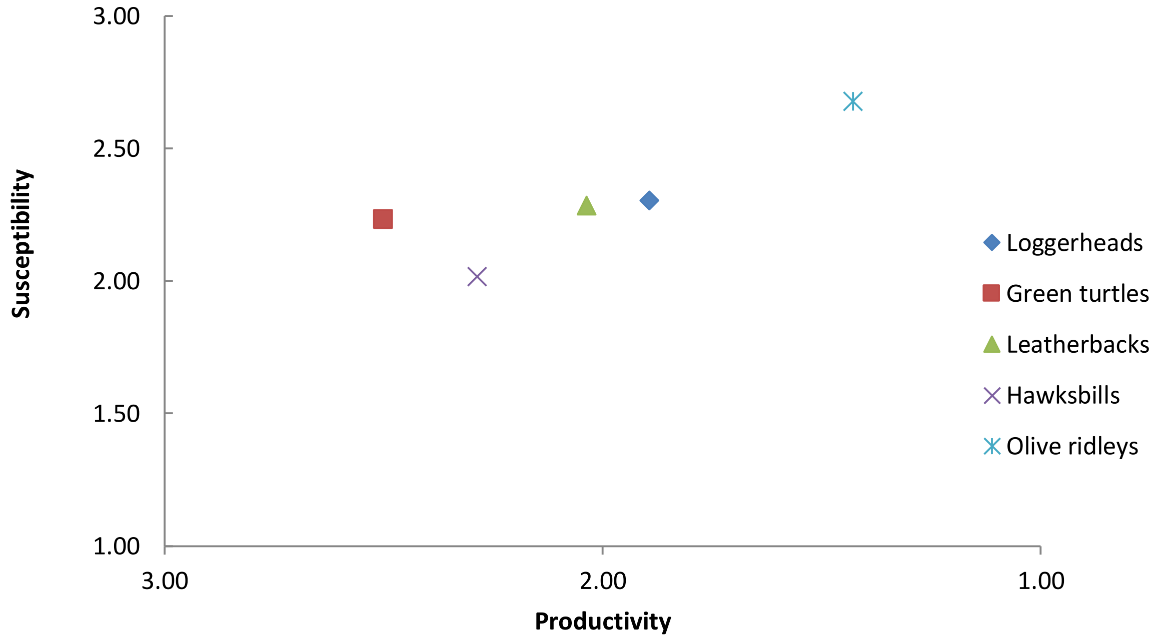


Figure H.4 Vulnerability graph of the gillnet fishery using susceptibility scores where the alternative threat score was included in the susceptibility analysis.

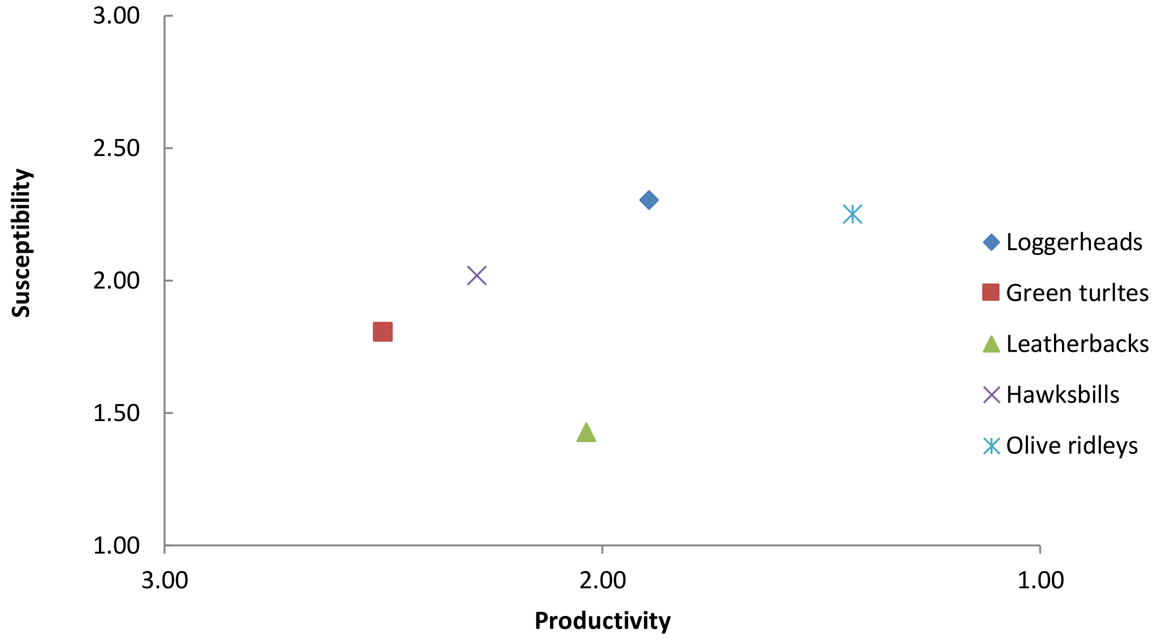


Figure H.5 Vulnerability graph of the beach seine fishery using susceptibility scores where the alternative threat score was included in the susceptibility analysis.

PSA graphs of vulnerability analysis excluding the alternative threat score

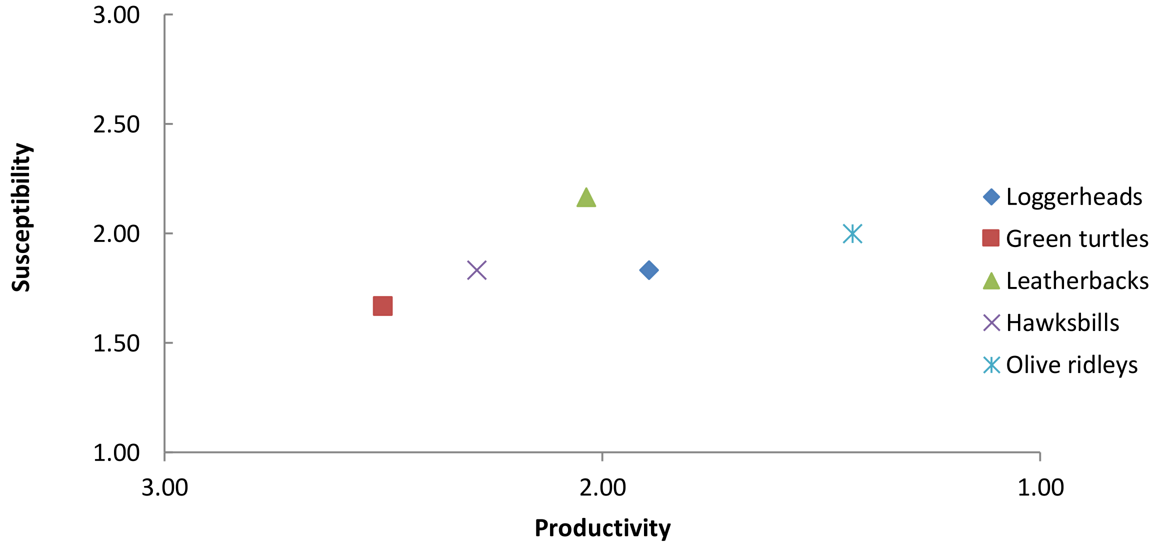


Figure H.6 Vulnerability graph of the longline fishery using susceptibility scores where the alternative threat score was excluded from the susceptibility analysis.

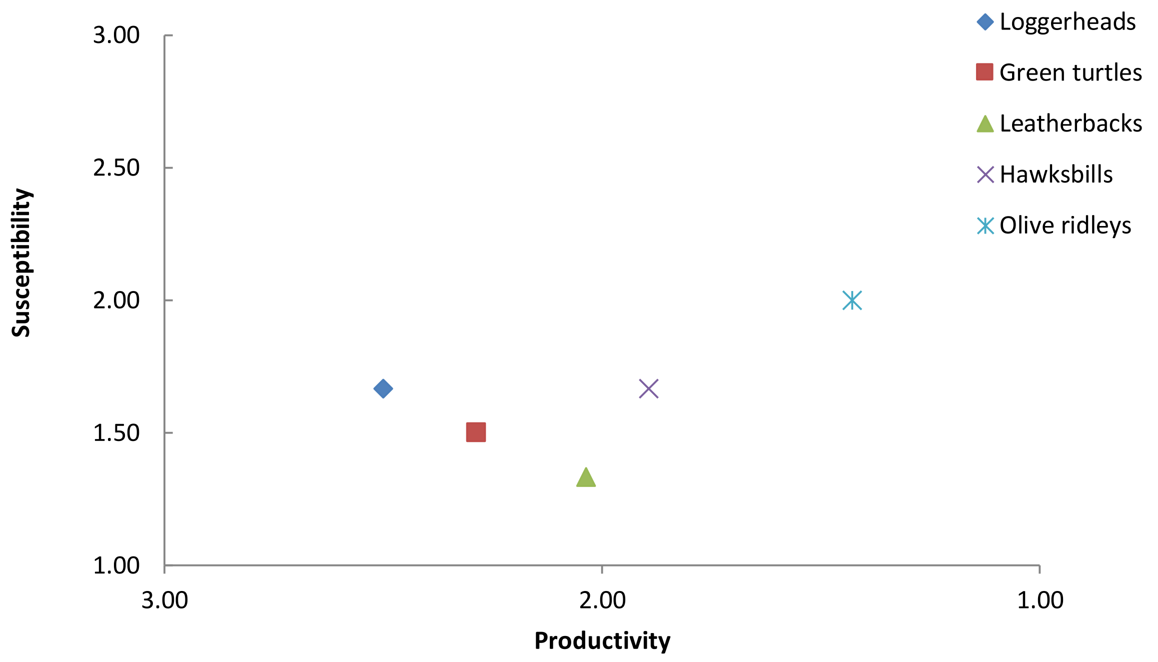


Figure H.7 Vulnerability graph of the purse seine fishery using susceptibility scores where the alternative threat score was excluded from the susceptibility analysis.

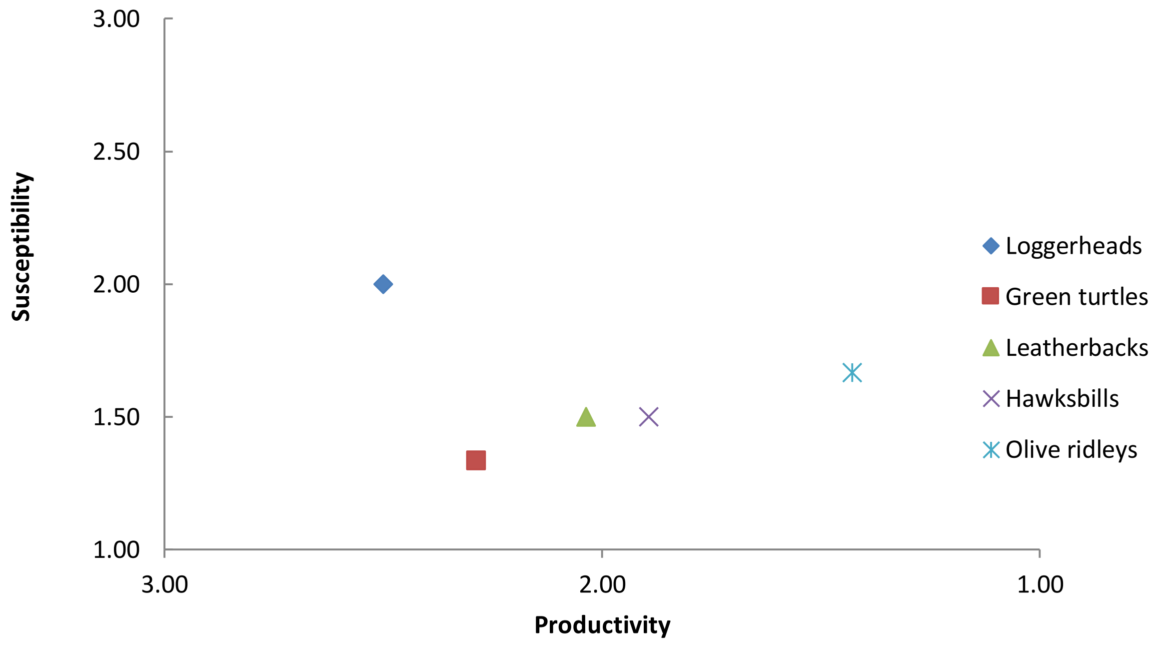


Figure H. 8 Vulnerability graph of the prawn trawl fishery using susceptibility scores where the alternative threat score was excluded from the susceptibility analysis.

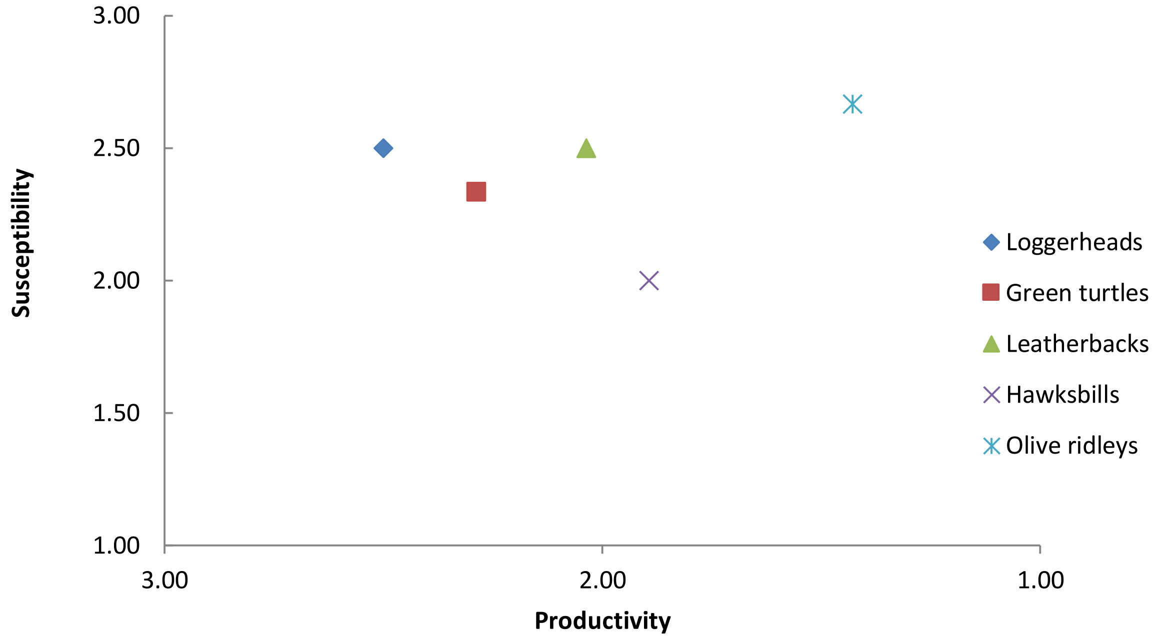


Figure H.9 Vulnerability graph of the gillnet fishery using susceptibility scores where the alternative threat score was excluded from the susceptibility analysis.

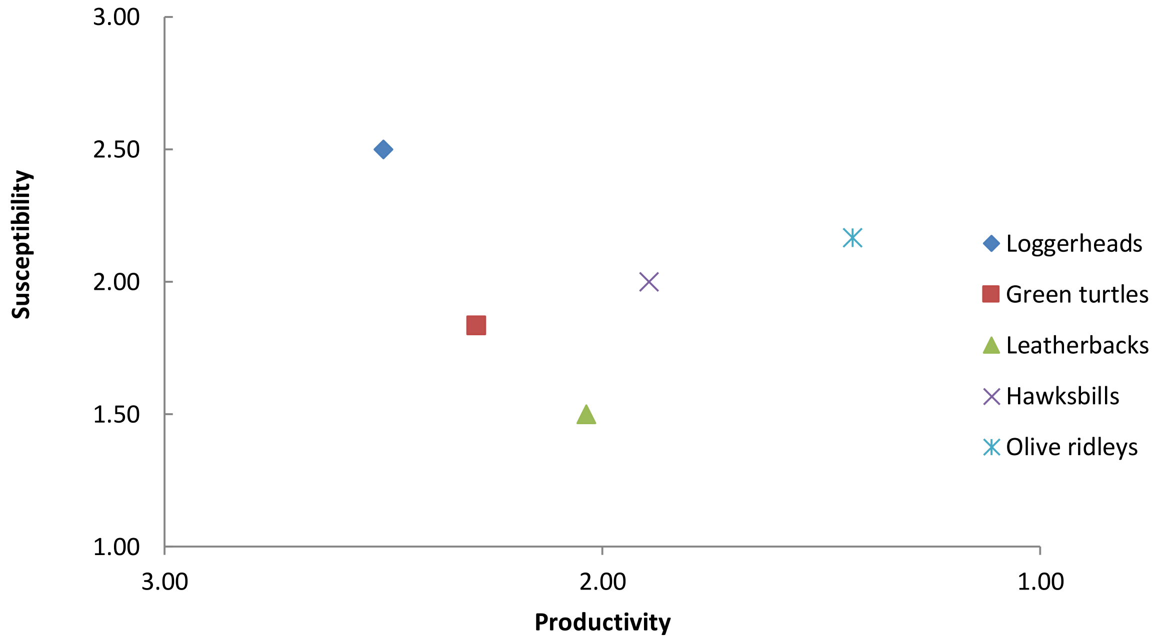


Figure H.10 Vulnerability graph of the beach seine fishery using susceptibility scores where the alternative threat score was excluded from the susceptibility analysis.

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