ECOLOGICAL RISK ASSESSMENT OF SEA TURTLES TO TUNA FISHING IN THE ICCAT REGION

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SUMMARY

Marine turtles spend the majority of their lives at sea; therefore understanding anthropogenic sources of mortality at sea is essential to assess population viability. This Ecological Risk Assessment assesses the risk to turtles from the impacts of tuna fishing in the ICCAT region. We used a Level 2 (semi-quantitative) assessment, within a Productivity-Susceptibility Analysis framework, at the Regional Management Unit (RMU) level; the assessment was hampered by significant data gaps and highly variable bycatch rate estimates. Bycatch rates were scaled to mean annual fishing effort, per RMU. ICCAT longline fishing poses the greater threat to turtles than purse seining. Loggerhead and leatherback turtles potentially encounter the most longline fishing effort (~300 million and >650 million hooks/yr, respectively). The east Atlantic olive ridley, the south Caribbean green turtle and SW Atlantic leatherback turtle RMUs were consistently among the most vulnerable from both gear types. Conversely, the west Atlantic olive ridely turtles showed lowest risk. Regions where turtles are at highest risk included S Caribbean and tropics (20°N-15°S, both gear types), and loggerhead turtles in the Mediterranean (longline only).

RÉSUMÉ

Les tortues marines passent la plupart de leurs vies en mer ; c'est pourquoi il est fondamental d'appréhender les sources anthropogéniques de la mortalité en mer afin d'évaluer la viabilité des populations. Cette évaluation des risques écologiques (ERA) évalue le risque pour les tortues provenant des impacts des pêcheries de thonidés dans la zone relevant de l'ICCAT. Nous avons utilisé une évaluation de niveau 2 (semi-quantitative), dans le cadre d'une analyse de productivité-susceptibilité (PSA), au niveau de l'unité de gestion régionale (RMU) ; l'évaluation a été entravée par des lacunes considérables dans les données et par les estimations des taux de prises accessoires fort variables. Les taux des prises accessoires ont été échelonnés à la moyenne de l'effort de pêche annuel, par RMU. La pêche à la palangre au sein de l'ICCAT constitue une plus grande menace pour les tortues que la pêche à la senne. Les tortues couannes et les tortues luth sont potentiellement confrontées à l'effort de pêche palangrier le plus grand (~300 millions et >650 millions hameçons/an, respectivement). La RMU de la tortue olivâtre de l'Atlantique Est, de la tortue verte du Sud des Caraïbes et de la tortue luth de l'Atlantique Sud-Ouest se trouvait de manière constante parmi les plus vulnérables avec les deux types d'engins. En revanche, la tortue olivâtre de l'Atlantique Ouest était exposée au risque le plus faible. Les régions où les tortues étaient les plus vulnérables incluaient le Sud des Caraïbes et les tropiques (20°N-15°S, deux types d'engins) et pour la tortue caouanne la Méditerranée (palangre seulement).

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RESUMEN

Las tortugas marinas pasan la mayor parte de sus vidas en el mar, por tanto, comprender las fuentes antropogénicas de la mortalidad en el mar resulta esencial para evaluar la viabilidad de la población. Esta evaluación del riesgo ecológico evalúa el riesgo para las tortugas procedente del impacto de la pesca de túnidos en la región de ICCAT. Se utilizó una evaluación de nivel 2 (semi-cuantitativa), dentro de un marco de análisis de susceptibilidad-productividad a nivel de unidad de ordenación regional (RMU); la evaluación se vio obstaculizada por las importantes lagunas de datos y por las estimaciones muy variables de tasa de captura fortuita. Se escalaron las tasas de captura fortuita al esfuerzo pesquero medio anual, por RMU. La pesca de palangre de ICCAT supone una amenaza mayor para las tortugas que la pesca de cerco. La tortuga boba y la tortuga laúd son potencialmente las que tienen más encuentros con el esfuerzo pesquero de palangre (~300 millones y >650 millones de anzuelos/año, respectivamente). Las RMU de las tortugas golfinas del Atlántico este, las tortuga verdes del Caribe sur y la tortugas laúd del Atlántico suroccidental fueron de forma continua las más vulnerables para ambos tipos de arte. Por el contrario, las tortugas golfinas del Atlántico oeste presentaban el riesgo más bajo. Las zonas en las que hay más peligro para los tortugas son: el Caribe meridional y tropical (20°N-15°S, ambos tipos de arte), y para las tortugas laúd el Mediterráneo (solo palangre).

KEYWORDS

By catch, ecological risk assessment (ERA), turtle, regional management unit (RMU), productivity-susceptibility analysis (PSA), vulnerability, gap analysis

1. Introduction

There are many reasons behind decreasing numbers of sea turtles. Anthropogenic sources include, *inter alia*, marine pollution (particularly plastic bags/debris), nesting habitat loss/degradation through tourism and coastal development, boat collisions, the direct take of adults, collecting of eggs by humans, and predation of eggs and hatchlings by invasive mammals on islands (FAO 2004). But because marine turtles spend the majority of their lives in coastal or pelagic waters, any cause of mortality at sea is a critical determinant to population viability and the largest of these is fisheries bycatch (Lewison et al. 2004, Wallace et al. 2011). However, turtle captures are generally an unwanted and unwelcome byproduct of fishing activities (termed bycatch) as evidenced by the large body of work directed at reducing turtle bycatch, particularly in large-scale fisheries (e.g. Gillman et al. 2006, Pacheco et al. 2011).

1.1 An Ecosystem Approach to Fisheries

This concept is also referred to as ecosystem-based fisheries management (Pikitch et al. 2004). The global trend towards implementing ecosystem-based approaches to fisheries management, rather than single species or target species-only approaches, has meant that responsibilities for assessing and managing non-target impacts of fishing fall increasingly to fishery management bodies. The Regional Fisheries Management Organisations (RFMOs) that manage tuna and tuna-like species began to acknowledge that responsibility, a move that was marked in the middle of the previous decade by the establishment of subsidiary bodies (known within ICCAT as Subcommittees) with a remit for assessing bycatch and other ecosystem considerations and making recommendations to higher RFMO bodies.

1.2 Tuna fishing and ICCAT

Tunas are amongst the highest-value marine species exploited by commercial fisheries for human consumption. The most abundant tunas are also highly migratory, moving between territorial waters of many nations and the high seas. The International Commission for the Conservation of Atlantic Tunas (ICCAT) was established to address the concerns of overfishing and the clear need to manage tuna stocks sustainably. Although a diversity of gear types are used to capture tunas, the two commonest gear types are considered here: longlining and purse seining. Gillnetting and other set-net techniques are known to capture turtles, often at very high rates (e.g. Lewison et al. 2007, Alessandro & Antonello 2010); however, these gear types are seldom used for tunas in the ICCAT region, gillnet effort represents less than 2% and are characterised by being extremely data-poor. At the Subcommittee on Ecosystems and By-catch (SC-ECO) meeting in July 2013, it was agreed that these gear types be disregarded for this analysis.

Longlining involves the deployment of a main line with buoys, radio beacons and other devices used to locate the lines at sea once deployed, keep the line at the correct setting depth, etc. As the main line is set, branchlines (typically 10-40 m long) with baited hooks (and sometimes with weights, lights or other devices attached) are clipped onto the mainline. The lines sink to target fishing depths which can range from <10 m to >500 m. Turtles are captured primarily in two ways – through becoming hooked when attempting to prey on baited hooks, or through entanglement when they swim across monofilament branchlines.

Purse seining involves setting a vertical wall of nets, with floats on the surface and weights on the bottom of the net to ensure the net hangs vertically from the surface. The net is deployed in a circle around a school of fish, and when encirclement is complete, the weighted, underwater section is pulled closed (or pursed – hence the name), and the net with the catch is hauled aboard. Setting is conducted on free-swimming schools and on fish aggregating devices (FADs), which can be anchored or drifting, natural or manmade. Turtles can become entangled in the netting used to construct FADs, where they may die of exposure or drown. They can also be accidentally captured alive during setting, although this seldom results in mortality (Clermont et al. 2012).

Other fishing methods, such as pole-and-line, rod-and-reel, trolling, etc. are more highly selective and there is no 'soak time' of gear in the water for extended periods, so any incidental captures can be dealt with immediately. These gear types are not as widely used, catch trivial biomass relative to the two major gear types, are not known to catch or interact with turtles to a significant level, and are not considered in this assessment.

1.3 Ecological Risk Assessment

Ecosystem impacts of fishing can be objectively evaluated within an Ecological Risk Assessment (ERA) framework (Arrizabalaga et al. 2011). Determining vulnerability to fisheries is straightforward for data-rich taxa, but much more challenging for bycatch taxa, where data collection is seldom a priority (Ormseth and Spencer 2011). Patrick et al. (2009) modified a previous framework to assess the sensitivity for these data-poor conditions (a PSA, or Productivity-Susceptibility Analysis). The intent of a PSA is to express productivity *P*, in terms of parameters such as age to maturity and fecundity of the population under investigation, in relation to the likelihood of being caught in a particular fishery (i.e. susceptibility, S). This likelihood is often expressed as the degree of spatial overlap between the fishery and the population, fishing intensity and gear selectivity (Ormseth and Spencer 2011).

The ERA with a PSA framework was developed for data poor environments, and can be conducted at one of three levels, depending on data quality; a qualitative Level 1 analysis, a semi-quantitative Level 2 analysis, and a fully quantitative, spatially explicit Level 3 analysis (Hobday et al. 2011, Tuck et al. 2011). A Level 3 ERA requires high-resolution spatio-temporal information on sea turtle bycatch in assessed fisheries (by national fleet or similar categorisation) and by gear type (longline, purse-seine etc.). It is also dependent on detailed population demographic information for sea turtles, preferably with annual census data, annual survival estimates for age/stage/sex classes, and knowledge on the spatio-temporal overlaps between fishing effort and turtle distribution by age/stage/sex classes. Previously, a Level 3 ERA was undertaken for seabirds under the auspices of ICCAT (Tuck et al. 2011). However, productivity and bycatch/susceptibility data for sea turtles in ICCAT (or elsewhere, e.g. Nel et al. 2013) are available only at the resolution or quality sufficient for a Level 2 ERA. Broadly speaking, a Level 2 ERA follows the methodology suggested by Milton (2001), modified by others (Arrizabalaga et al. 2011, Ormseth and Spencer 2011), using indicators of population productivity and susceptibility to capture in different fisheries.

This assessment was requested by the SC-ECO of the ICCAT Standing Committee on Research and Statistics (SCRS) to address ICCAT recommendation 10-09 which requires that the SCRS initiate an assessment of the impact of the incidental catch of sea turtles resulting from ICCAT fisheries. Terms of reference were provided by the ICCAT secretariat to conduct an ERA on the impact of ICCAT fisheries on sea turtles, at the level of Regional Management Units (RMUs) and based on information previously collated.

It should be noted that this is a relative assessment of risk. Estimated values should not be treated as firm estimates of likely actual annual mortality or absolute scores of risk. The primary objective is to identify the populations most at risk and regions where threats are likely to be highest, a fundamentally comparative and relative objective. This approach was informed by the quality and quantity of data. Fishing data housed by ICCAT are, for the most, detailed, comprehensive and contain multiple attributes suitable for stratifying analyses or using in predictive models. To stratify impacts or risks to turtles, self-evidently some information of turtle capture rates/probabilities per stratum is required. In some regions turtle interactions with different gear types are documented, which would facilitate detailed analyses of risk relative to season, hook-type, bait-type, setting depth, etc. However, for the great majority of RMUs included in this assessment, and for the majority of fishing fleets, interaction data are simply not available or only at levels of gear type and broad geographical region. Further, for all turtles, the demographic information do not exist that would be required to conduct detailed analyses of risk which account for different catchability/exposure estimates by age, sex or colony within an RMU. As this analysis is the first attempt to assess the risk of all ICCAT fisheries to all turtles, we chose to retain full comparability between regions and RMUs, so as to highlight areas where more observer effort, research or measures to mitigate potentially significant capture can be directed. It will also, hopefully, stimulate further efforts, provision of additional data, and more detailed assessments in future.

1.4 Sea turtles and regional management units

There are seven species of sea turtles most of which are distributed in tropical and sub-tropical waters. All are listed as threatened on the IUCN Red list **(Table 1).** Six species either nest or spend part of their life cycle in Atlantic Ocean.

The concept of RMUs was developed by Wallace et al. (2010a), because resolving threats to widely distributed marine megafauna, such as sea turtles requires definition of the geographic distributions of both the threats as well as the population unit(s) of interest. The RMU framework is used as an approach to deal with the threats to sea turtles, which although are widely distributed, exhibit extremely high natal site fidelity and, as shown by genetic analysis, show very little reproductive interaction between regional populations. In total 58 RMUs have been identified globally (Wallace et al. 2010a). It provides a useful tool for *inter alia* identifying data gaps, evaluating conservation status of sea turtles and providing guidance to management initiatives. We recognized 22 RMUs within the Atlantic Ocean, belonging to six species (**Table 2**).

The Atlantic Ocean is home to some of the largest and some of the smallest RMUs in the world (Wallace et al. 2010a). It hosts the largest populations (RMUs) of the critically endangered leatherback turtle (in Gabon) and the fastest growing RMUs (in the Northwest Atlantic). The east Atlantic populations of both hawksbill and olive ridley turtles, and the leatherback turtles in the south Atlantic (both east and west) are amongst the smallest RMUs in the world and are likely to be the most susceptible to fishing pressures; the South Atlantic leatherback turtles are at risk from high seas fisheries as they undertake trans-Atlantic migrations (Marcovaldi et al. 1999) and the hawksbills and olive ridley turtles are at risk from coastal fisheries as they have a more coastal distribution (Wallace et al. 2010a). Kemp's ridley is endemic to the Atlantic Ocean, occurring in the Gulf of Mexico and the east coast of the U.S.A. Its population is moderate in size and its mostly coastal distribution will put it at most risk from coastal fisheries (such as gillnetting). The green and loggerhead turtles of the Atlantic and Mediterranean are generally moderate in size (i.e. 1000 – 5000 adult females nesting per annum).

1.5 Atlantic Ocean RMU's Summary

1.5.1 Loggerhead turtle (Caretta caretta)

1. Atlantic northwest (Cc-AtNW)
   The loggerhead turtle nesting population in the northwest Atlantic Florida, North Carolina and Bahamas) consists of approximately 17 500 nesting females (Ehrhart, Bagley and Redfoot 2003). Depending on the region, this population is considered stable or increasing. Increased human presence at nesting beaches threatens future nesting events mainly because of an increase in artificial lighting changed sand deposition regimes (Antworth et al. 2006). Furthermore, anthropogenic food provision has led to unnaturally high numbers of nest predators (particularly raccoons). Besides incidental catch in fisheries (also dredging), other offshore threats include pollution and power plant entrapment (Brazner and MacMillan 2008).
2. Atlantic northeast (Cc-AtNE)
   The Cape Verde Islands has a loggerhead turtle nesting population of approximately 1000 individuals (Ehrhart, Bagley and Redfoot 2003). Current trends in the number of nests suggest that this population is decreasing. This is mainly because of habitat destruction (coastal development) as well as high levels of illegal harvesting of eggs and nesting females (Marco et al. 2011). Incidental and targeted catch of sea turtles are the main offshore threats (Bolten et al. 2000, Mejuto 2008).
3. The Mediterranean (Cc-AtMed)
   The loggerhead turtle nesting population in the Mediterranean spans the coasts of Turkey, Greece and Cyprus and comprises 2280 - 2787 individuals (Broderick et al. 2002). Currently this population is stable, but the destruction of nesting habitat is a looming threat (Margaritoulis 2005). Marine pollution is a major threat to this population. Loggerhead turtles are often caught incidentally in commercial fisheries that operate in this sea and they are vulnerable to boat strikes (Casale et al. 2008).

4. Atlantic southwest (Cc-AtSW)
   The loggerhead turtle nesting population in Brazil is increasing with a mean annual number of 1237 nesting females (Marcovaldi & Chaloupka, 2007). Nesting beaches are under severe pressure from coastal development (Baptistotte et al. 2003). The greatest offshore threat is incidental capture in both coastal and pelagic fisheries (Kotas et al. 2004, Sales et al. 2008).
5. Indian southwest (Cc-InSW)
   The loggerhead turtle nesting population in the southwest Indian Ocean is shared with Mozambique. This population is showing an increasing trend with a mean annual 371 nesting females (Nel et al. 2013). Because the nesting grounds fall within a world heritage site, harvesting of eggs and nesting females have for the most part been eliminated. Major threats to this population occur offshore in the form of incidental fisheries bycatch (de Wet 2012).

1.5.2 Green turtle (Chelonia mydas)

6. Atlantic northwest (Cm-AtNW)
   The green turtle nesting population in the northwest Atlantic Ocean is increasing. There are an estimated 17 402 - 37 290 individuals that nest in this region annually (Troeng & Rankin 2005). The major land-based threat in Florida is coastal development whereas Costa Rica has the problem of direct harvesting of eggs and nesting females (Troeng & Rankin 2005). Legal harvest of eggs must remain controlled to ensure the continuous increase of this population. Offshore threats include incidental catch in commercial fisheries as well as various forms of pollution (Beerkircher et al. 2004).
7. Atlantic south Caribbean (Cm-AtScar)
   In Suriname, there are between 267 and 1816 green turtle females that nest annually (MTSG 2004). This population is considered to be increasing.
8. Atlantic south Central (Cm-AtSCen)
   The green turtle population is at Ascension Island (1650-3000 nesting females per annum) is increasing whereas the population at Bioko is considered stable (415 - 560; Radar et al. 2006). Main threats at Bioko include coastal development, destroying nesting habitat and increasing light pollution, as well as the illegal harvesting of eggs and females (Radar et al. 2006). Incidental catch in commercial fisheries and targeted catch by subsistence fishermen (Bioko) are of great concern (Tomas et al. 2010). Further offshore, threats include plastic ingestion and ghost fishing (entanglement in lost/discarded fishing gear) (Tomas et al. 2010)).
9. Atlantic southwest (Cm-AtSW)
   In the southwest Atlantic (Brazil and associated islands), there are between 600 - 1200 green turtle females that nest annually (Almeida et al. 2011). This population is considered to be stable, but incidental catch in commercial fisheries an immediate concern (Gallo et al. 2006).
10. Atlantic east (Cm-AtE)
   There are approximately 2000 females that nest on the beaches of Guinea-Bissau annually and this population is thought to be increasing (Catry et al. 2009). However poaching of eggs and nesting females is still a problem (Catry et al. 2002). Also, nests are vulnerable to density dependent mortality. The main offshore threat is incidental capture in fisheries.
11. Indian southwest (Cm-InSW)
   There is a very large population of green turtles that nest on islands in the south western Indian Ocean. The number of females that nest annually is increasing at both Europa and Tromelin islands. The main natural source of mortality is egg predation. The main non-natural threats are the fisheries throughout the Mozambique Channel (Bourjea et al. 2008).
12. Mediterranean (Cm-Med)
   The main nesting activity of green turtles in the Mediterranean occurs along the coastlines of Turkey. This population is declining, with approximately 400 individuals nesting annually. Major threats to this population include incidental catch in fisheries, boat strikes as well as entanglement in marine debris and ghost fishing (Casale 2011).

1.5.3 Leatherback turtle (Dermochelys coriacea)

13. Atlantic Northwest (Dc-AtNW)
   The leatherback turtle nesting population throughout the Caribbean is increasing, with an estimated 2900 - 8500 individuals nesting per annum (Fosette et al. 2008). Terrestrial threats include coastal development and the increase of artificial light pollution (Dutton et al. 2005). In addition to incidental catch in fisheries, other offshore threats include pollution and power plant entrapment.

14. Atlantic south, divided into two sub-populations: 14a (Atlantic southeast) and 14b (Atlantic southwest)
    14a. Atlantic southeast (Dc-AtSE)
    This population is considered to be the largest leatherback rookery in the world. It is estimated that there are between 10 000 and 25 000 leatherback turtle females that nest along the coast of west Central Africa (Fosette et al. 2008). The trend of this population is currently unknown. Nevertheless, the most significant beach-based threat is illegal egg harvesting (Verhage et al. 2006). Density-dependent nest destruction (where turtles inadvertently dig up other turtle nests because of space constraints) is common. Threats at sea include incidental catch in fisheries, ghost fishing and entanglement in marine debris (Weir et al. 2007, Mejuto et al. 2008).
    14b. Atlantic southwest (Dc-AtSW)
    This is a very small nesting population of leatherback turtles and population trends are largely unknown. However, threats include nest erosion, fisheries bycatch as well as oil-related incidences (Pacheco et al. 2011).
15. Indian southwest (Dc-InSW)
    Approximately 80 leatherback females nest in the south western Indian Ocean annually (Nel et al. 2013). Although this population is small and stable, it is under the protection authority of a world heritage site. The main beach-based threat to this population is nest erosion, particularly in the face of climate change. Offshore threats largely consist of incidental capture in coastal and pelagic fisheries (de Wet, 2012).

1.5.4 Hawksbill turtle (*Eretmochelys imbricata)*

16. Atlantic west Caribbean (Ei-AtWCar)
    Approximately 5000 hawksbill turtles nest throughout the Caribbean, including the beaches of Florida (Meylan 1999). Although this population is showing an increasing trend, habitat destruction because of coastal development is a growing concern (Beggs et al. 2007). Bycatch, ghost fishing and entanglement in marine debris are the main threats offshore (Meylan 1999).
17. Atlantic east (Ei-AtE)
    There is a very small population of hawksbill turtles at Bioko and Guinea-Bissau, averaging 10 nesting females per year (Tomas et al. 2010). This population is under severe pressure from egg collecting. At-sea threats include incidental catch in commercial fisheries as well as targeted catch by subsistence fishermen (Catry et al. 2009). Furthermore, this population is at risk from oil exploration activities (Tomas et al. 2010).
18. Atlantic southwest (Ei-AtSW)
    The hawksbill turtle population in Brazil and surrounds is increasing (350 – 585 nesting females per annum; Mortimer and Donnelley 2007). However, coastal development together with light pollution continues to threaten the recovery of this population. The main offshore threats include incidental catch in fisheries and various forms of pollution (Mortimer and Donnelley 2007).

1.5.5 Olive ridley turtle (Lepidochelys olivacea)

19. Atlantic west (Lo-AtW)
    The Olive ridley turtle population in the western Atlantic is increasing, thanks to controlled egg collection and conservation programs. Approximately 1716 – 3257 females nest throughout Brazil, Suriname, Guinea and Venezuela (Kelle et al. 2009). The main terrestrial threat is increasing human presence at nesting beaches, with factors such as coastal development, artificial lighting and vehicles on beaches threatening this population (da Silva et al. 2007). Incidental bycatch in the shrimp trawl fishery is still a concern.
20. Atlantic east (Lo-AtE)
    The trends of the small (19 – 43) nesting population of Olive ridley turtles in West Africa are unknown. These turtles are exploited for eggs and meat and their nests are frequently raided by feral dogs (Weir et al. 2007). Offshore threats include incidental catch in commercial fisheries as well as targeted catch by subsistence fishermen (Weir et al. 2007).

1.5.6 Kemp's ridley turtle (Lepidochelys kempii)

21. Atlantic northwest (Lk-AtNW)
    The Kemp's ridley turtle population in the northwest Atlantic is increasing and it is estimated that between 7000 and 8000 females nest here annually (Crowder and Heppel 2011). However, egg poaching is still a problem. Oil exploration, oil pollution and incidental bycatch in shrimp trawl fisheries are the main at-sea threats (Lewison et al. 2003).

2. Methods

The main aim of this Level 2 ERA for ICCAT fisheries was to estimate likely rates of turtle mortality from fishing operations, per RMU, relate those to each RMU's demographic parameters in a statistical model and estimate the likely relative scale of impacts to each turtle RMU by gear type. Broadly speaking, this approach involved several steps:

1. Identify turtle RMU at-sea distributions
2. Quantify turtle Bycatch Per Unit Effort (BPUE) rates per RMU and per gear type
3. Quantify the level of fishing effort to which each RMU is exposed
4. Extrapolate from BPUE and effort estimates to calculate the number of turtles killed per RMU and per gear type
5. Include mortality estimates in a PSA to estimate the relative risk faced by each RMU in the ICCAT region

All data used for the generation of this assessment, including references, have been provided to the ICCAT secretariat as follows:

- Appendix 1 – All raw BPUE data and calculations of estimated total bycatch and mortality per RMU – Excel spreadsheet.
- Appendix 2 – Productivity and susceptibility analysis matrix, including productivity and susceptibility parameters and criteria – Excel spreadsheet.
- Appendix 3 – Satellite metadata – Excel spreadsheet.
- Folder – GIS (Global Information System) containing all files and metadata used to create the effort and RMU overlay maps.

2.1 Turtle biology, bycatch and fisheries effort data

An extensive literature review was conducted on turtle biology and demographics, and some 90 references were used to populate the productivity matrix (see Appendix 2 for references used). Where the published literature was older than 10 years, demographic values were confirmed using global databases, in particular the State of the Word's sea Turtles (SWOT), and the most recent IUCN red listing assessments. Extensive consultation was also carried out with turtle experts to refine and expand on published knowledge. Datasets from >2000 satellite tracked individuals in the Atlantic Ocean were obtained from Seaturtle.org (www.seaturtle.org). The RMU boundaries were obtained as shapefiles from SWOT (www.seamap.env.duke.edu/swot) which are freely available online.

Data on turtle interactions and capture rates in ICCAT fisheries were provided to the SC-ECO group, in response to a specific call for such data. In addition to other data sourced, a comprehensive review of published and grey literature, on catch rates and related datasets, was undertaken for ICCAT fisheries (Coelho et al. 2012). This covered >170 publications and summarised the major findings in terms of turtle interactions in the ICCAT convention area, by gear type: longline, purse seine and drift nets. The authors also included other non-ICCAT fisheries in the Atlantic Ocean, such as artisanal surface longlining and shrimp trawl fishery. The bulk of the information, however, relates to longline fishing. Coelho et al. (2012) provide tabulated summaries for all reported turtle bycatch per unit effort (BPUE) data from the northwest and northeast Atlantic (including the mid-Atlantic continental shelf and western Azores), southwest and southeast Atlantic, and the Mediterranean Sea. For purse seine turtle interactions, we found no data other than that reported in Clermont et al. (2012), who reviewed EU purse seine fishery in the Atlantic Ocean (the main purse seine fishery in ICCAT). Clermont et al. (2012) also attempted a scaling up from observed catch to the total purse seine effort and we have therefore used their results in this analysis.

We used longline and purse seine effort data provided by ICCAT, stratified by flag, month and 5°x5° and 1°x1° resolution, respectively. Some publications report BPUE by target species or target setting depth, but this was inconsistent and absent entirely from large areas of ICCAT longline effort. Longline effort was stratified by depth for use in the shark ERA done by ICCAT, but that dataset was not made available for this study. Furthermore, there were too few studies reporting turtle BPUE estimates with associated setting depth or target species. As a result we could not account for setting depth or exposure period for the whole of the convention area in this risk assessment. Appendix 1 lists all available BPUE data.

Most papers report nominal BPUEs, but where standardized BPUE values were available they were included in preference to nominal data. We discarded all "unknown" or "sea-turtle nei" data from analyses. Observer, logbook and scientific experimental data were used in the analysis. The most recent distribution data available was used to assign reported turtle BPUE to a particular RMU. For instance it has been determined, using mtDNA sequence analyses, that most juvenile loggerhead turtles caught off the Azores belong to the Northwest Atlantic RMU and probably not to the Northeast Atlantic RMU with which it overlaps, and BPUEs where thus assigned accordingly (Bolten et al. 1998, Bolten 2003). Only when this proved impossible, based on the catch information provided and defined turtle distributions, was a reported turtle BPUE assigned separately to both overlapping RMUs. Some of the reported BPUEs used in this assessment are drawn from studies with overlapping years, and therefore there is a small degree of non-independence; however without access to the raw data bycatch data it was impossible to control for in the analyses.

2.2 Productivity

Productivity information was obtained for sea turtles species breeding on beaches facing the Atlantic Ocean (**Table 3a**). Ten productivity parameters were identified (nine in the original terms of reference, to which we added one), and sufficient information was available for nine of these (**Table 3a**). Parameters were assigned scores (1-3) with three as the highest productivity. The number of breeding females was used as a proxy for population size, which is standard procedure when dealing with sea turtles because little to no information is available on any other stage classes or male turtles; these values were categorized to assign scores according to **Table 3b.** The remigration interval refers to the number of years between breeding attempts.

2.3 Susceptibility

The susceptibility analysis focussed on the horizontal overlap of turtles and fisheries operations, as well as variables that may influence interactions between turtles and different gear types. Nine parameters were identified for inclusion in the analysis, but data were available only for two of these: spatial overlap of each RMU with the ICCAT region and an estimate of the impact of mortality from bycatch (for two gear types – longlining and purse seining). However, these are arguably the two most important parameters to assess risk directly (e.g. Small et al. 2013).

2.3.1 Excluded parameters

Although the ERA methodology is precautionary and calls for data gaps to be scored high (Hobday et al. 2011), where insufficient data exist for all RMUs, the effect of scoring everything high is the same as excluding it, as it contributes no discriminatory power to the assessment. Seven of the original parameters could not be assessed because data were either not available, or too few to sensibly include in this analysis. Also, the relevance of some parameters varies between gear types.

A key deficiency is detailed spatio-temporal turtle distributions. Ideally, raw (or similar) data from individual-based tracking studies with robust sample sizes, from each RMU, preferably with good coverage of the rookeries within each RMU, would be used (c.f. seabird ERA for ICCAT, Tuck et al. (2011) and references therein). However, there is insufficient tracking data for most RMUs. Further, most tracking data are proprietary, and there were insufficient time and resources to establish the collaborative agreements with all (or most of the important) data owners such as would be required to access raw tracking data. This meant that the parameters 'geographical concentration' and 'temporal overlap' with fishing effort could not be assessed. Although some fine-scale information on turtle spatio-temporal habitat use in the ICCAT region is available (e.g. Pons et al. 2010), it is unlikely that sufficient data exist to assess temporal overlap at the scales required for this assessment.

A second key deficiency is in the kinds of data collected by fisheries. The parameter 'vertical overlap' relates to the depth at which gear is set, but ICCAT databases do not contain these data. Retention of turtles in gear, length of set (or 'soak time') and depth of hooking (longlining, e.g. Chaloupka et al. (2004)) or damage from gear interactions (all gear types) are poorly quantified or poorly reported and we found no useful information to inform these parameters, either in collated data and papers provided by the ICCAT secretariat or from other sources. Soak time is likely to be an important discriminating factor in shallow- and deep-set longlines and although there are localized studies looking at these parameters (e.g. Gilman et al. 2006, Gardner et al. 2008, Lewinson et al. 2013) there is not enough bycatch information data for the ICCAT convention area to stratify our analysis at that level. However, the parameters 'vertical overlap', 'retention in gear', 'length of set' and 'hooking depth/physical damage' inform why a particular bycatch/mortality rate occurs, and provide insights into how that rate might be reduced. They could also be used as proxies if a bycatch rate was not available. But in a coarse-scale ERA such as this they become less important than estimating likely annual mortality.

A gap common to several of these non-assessed parameters is the rate or scale of mortality once a live turtle has been caught, disentangled/removed from gear and released. This is generally referred to as post-release mortality, and is essential to assess fishing impacts properly, but is virtually unknown (Hays 2003, Chaloupka et al. 2004, Swimmer et al. 2006).

2.3.2 Leslie matrices and natural mortality

A critical component of a PSA is assessing the impacts of fishing relative to natural (and other sources of) mortality. The original approach envisaged for this ERA was to construct RMU-specific Leslie population matrices, which would be used to estimate natural annual mortality and to compare this to estimated mortality from ICCAT fisheries. However, there are no age-structured demographic data for RMUs in the Atlantic Ocean (M. Chaloupka pers. comm., Bjorndal et al. 2011). Thus no age-structured demographic tables can be generated. Indeed very few Leslie matrices or similar stage-based demographic models have ever been produced for sea turtles (Chaloupka & Musick 1997, Crouse et al. 1987, Crowder et al. 1994, Heppell et al. 2005, National Research Council 2010). Several authors have attempted to estimate mortality in sea turtles (Chaloupka & Limpus 2005, Troëng & Chaloupka 2007) but existing models are still not capable of explaining adequately the known patterns from all life history stages (Hamann et al. 2010)

2.3.3 Included parameters

The overlap parameter was calculated as the number of 2.5^o squares covered by an RMU's distribution, as indicated from satellite tracking studies, as a proportion of all squares (2000) in the ICCAT region (for calculations Appendix 2). Datasets from >2000 tracked individuals were digitised, but due to proprietary data concerns, each individual's tracks were mapped as presence/absence in each square (see Appendix 3 for metadata). Squares were shaded more intensely where tracks from multiple individuals overlapped. Self-evidently the more tracked individuals per RMU, the greater the probability that the complete range of the RMU is described; there is likely to be greater heterogeneity in the distributions than indicated in this exercise (e.g. Pons et al. 2010). For example, areas within the RMU boundaries may not be used, and additional tracking studies are likely to both expand current boundaries and indicate areas where turtles avoid. This parameter was scored low if tracks covered <50 squares (equivalent to ~2.5% of the ICCAT region), medium if >50 but <100 and high if >100 squares (~5% of the ICCAT region; **Table 4**).

The number of tracked individuals varied appreciably between RMUs, from 0-250. To account for this we added a new parameter – the number of tracks used to assess the overlap parameter. This provides a basic indication of the confidence in the score given to each RMU overlap estimate and this parameter was scored as low if the number of tags in the RMU was <5 tags, medium if >5 but < 30 and high if >30 tags for an RMU (**Table 4**).

The parameter 'bycatch mortality relative to natural mortality' requires two things, a bycatch mortality estimate and an estimate of the impact that this will have on an RMU. For the first component (bycatch mortality), all available data sources were mined in an effort to establish estimates of turtle capture rates per gear type and per RMU. This parameter was based largely on reported numbers of turtles captured alive, one of the original susceptibility parameters. Turtles captures alive was not retained as a separate parameter for two reasons. First because it forms a core component of the data used to estimate annual bycatch for both longlining and purse seining (most studies report catch, with relatively few reporting proportion of catch that was dead or died), and scoring a factor in multiple parameters should be avoided. Second because this is only of value when post-capture mortality rates are available, but they are not (Chaloupka et al. 2004). BPUE was calculated as numbers caught per 1000 hooks (longlining) and per set (purse seining) and assigned to RMUs. From this we estimated mean and standard deviations of reported BPUEs for each RMU. RMU boundaries were overlaid onto heat maps of ICCAT longline fishing effort to estimate the likely annual scale of effort to data which each RMU is exposed. To overcome inter-annual variability in fishing effort and to provide a general likelihood of the level of effort which a given RMU encounters annually, we used the 10-year mean of reported fishing effort data from 2000-2009. Effort data were mapped at 5°x5° resolution for longline and at 1°x1° for purse seine, using mapping software (see Folder GIS for files used to create all maps). We then scaled the BPUE estimates to total effort to yield estimates of total annual catch per RMU and per gear type. Coelho et al. (2012) also reported data on catch:mortality ratios for longline. In most instances total catch is recorded, but the fate of captured turtles (dead or alive) is reported less frequently. Appendix 1 shows the available catch: mortality ratios for each RMU. Because so many RMUs were data-deficient for this aspect, we assumed that species-specific factors were most likely to influence the relationship between numbers captured and numbers that died. So as not to bias areas with few data we averaged catch: mortality ratios for each species and used those to convert captures into total annual mortalities per RMU.

Without Leslie matrices or other models to estimate the relative impact of bycatch mortality on each RMU, we chose to scale the impact of bycatch mortality relative to another demographic parameter. Intra-specific catch was assumed to occur in proportion to abundance of age- and sex-classes, i.e. there is no *a priori* reason to suggest captures are biased towards males/females or certain age-classes (but see Crowder et al. 1994). Bycatch mortality (assumed to encompass all classes) was compared to the only turtle demographic parameter for which reasonably robust estimates exist – numbers of breeding females. This is appropriate because turtles are polygynandrous, so females are the limiting resource for reproduction. Mortality per RMU was converted to an estimate of adult female mortality using the following heuristics: females constitute half the population, and adults constitute 10% of the population. Thus if total mortality was likely to include <5% of the female population, we scored the parameter as 1 (low) for the RMU. For simplicity, rather than perform those calculations for each RMU, they are equivalent to the following scale: if the numerical value of a mortality estimate (from all age, stage and sex classes) is ≤30% of the adult female population, then ≤5% of adult females killed. Therefore we scored this parameter as total bycatch relative to the adult female population as follows: 1≤30%, 30%<2>100%, or 3>100%.

As with Productivity parameters, Susceptibility scores were rated (1-3), however unlike the former, the greatest risk to each RMU was scored a three. Thus small, less-productive RMUs (most vulnerable to fishing impacts) scored lowest on the productivity scale, but the scale of susceptibility worked in the opposite direction. This was addressed when calculating relative vulnerability.

2.3.4 Weighting parameters

Productivity parameters were weighted based on our confidence in the available data and in the degree to which they influence population resilience. Rookery size (either through direct female or track counts) is probably the most reliable and important parameter, and hence was weighted the heaviest (3). The trend data is similarly robust (many of the land-based monitoring programmes have been operating for 10 or more years) and if a population is decreasing, any bycatch mortality will be deleterious, so this was weighted next highest (2). Hatching and emergence success provide direct measures of productivity but are inconsistently reported, so were weighted down (0.5). All other metrics were unweighted. Susceptibility criteria were weighted equally (**Table 4**) for all gear types assessed.

2.3.5 Calculating relative vulnerability

Both Productivity and Susceptibility scores were summed and rescaled to range from 1-3. Relative vulnerability V, was calculated as

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

where P is the productivity score for the RMU and S is the susceptibility score for the relevant RMU and gear type. In addition, Euclidean distances between RMU's V provided a quantitative measure of relative vulnerability (Ormseth and Spencer, 2011).

3. Results

3.1 Productivity

P scores ranged from 1.2-2.6 with a mean score of 2.0 (**Table 5**). The hawksbill turtle RMUs had the most extreme distribution within this analysis, with East Atlantic RMU (Ei-AtE) scoring lowest (score = 1.2) but the other two RMUs, from the Caribbean (Ei-AtWCar) and Southwest Atlantic (Ei-AtSW) ranked second with tied scores of 2.55. Ei-AtE was also noticeable as the only RMU separated from other scores by 0.3 points. This was mostly due to the conservative scoring, due to very low data availability. The most productive (score = 2.6) was the green turtle RMU from the SW Indian (Cm-InSW), which has a distribution relatively marginal to the Atlantic Ocean. Aside from that, four of the six remaining (and Atlantic-specific) green turtle RMUs were in the lower half of the ranked scores = 1.7. Considering species-specific rather than RMU-specific scores, Kemp's ridley scored highest (2.3), but as it constitutes just one RMU it is not an average score. The loggerhead turtle mean score was the only species to average above 2 (mean score = 2.01).

3.2 Susceptibility

3.2.1 Longlining

BPUEs were not reported for five of the 22 RMUs in the Atlantic Ocean, so for the bycatch risk parameter these RMUs were scored 3, the highest risk category. The mean for all RMUs which had at least one BPUE reported, was 0.283 turtles/1000 hooks. All loggerhead turtle RMUs had BPUEs reported, with the highest mean catch rates from the southwest Atlantic (1.182 turtles/1000 hooks). Loggerheads also had, on average, BPUEs 2-3 times higher than reported means for green or leatherback turtles. The enormous geographic range of leatherback turtles meant that for the three RMUs endemic to the Atlantic Ocean, the mean annual effort to which they are exposed dwarfs that of any other Atlantic RMU. The highest ranked was the Atlantic Northwest RMU (Dc-AtNW), which encountered potentially as much as 270 million longline hooks annually; the other two encountered effort approaching 200 million hooks/year. It's noteworthy that two of three RMUs which straddle the Atlantic and Indian oceans (Cc-InSW and Dc-InSW), and which in the Atlantic Ocean are therefore confined to the SE Atlantic, had reported BPUEs that were an order of magnitude lower than their conspecifics elsewhere in the Atlantic. The third straddling RMU (Cm-InSW) has similarly very low BPUE (0.026 turtles/1000 hooks), which is less than half the next lowest green turtle RMU's BPUE; this pattern of significantly lower BPUEs in the SE Atlantic merits closer investigation. Some possible explanations include that turtle species occur at much lower densities in this region than elsewhere. This is supported by satellite tracking; to date none of the greet turtles tagged in the south Western Indian Ocean has entered the Atlantic Ocean. However, genetic analysis has confirmed a link with the Atlantic Ocean and hence this RMU is included in this analysis (Bourjea et al. 2007). Alternatively there may be differences in foraging behaviour rendering them less susceptible to bycatch, or there is under-reporting of turtle bycatch from this region.

When the mean of reported BPUEs per RMU were scaled up to the longline effort to which each RMU is estimated to be exposed, seven of the 16 RMUs for which an estimate could be made had captures in the 10s of thousands annually. Excluding the outliers from the SE Atlantic, the three leatherback turtle RMUs totalled an estimated 165,000 individuals captured on longlines every year (mean ~55,000/RMU/year) and loggerhead turtle RMUs totalled ~200,000 captures/year (mean ~50,000/RMU/year) (**Figures 1-6**).

Reported percentages of captured turtles that died immediately varied considerably but were generally around 5%, i.e. ~5% of reported captures were animals that were dead on hauling or died before being released. Notable exceptions were twofold. First, there were no ratios reported for hawksbill turtles. Second, the two ridley turtle species had much higher immediate mortality rates than other genera (18% for olive ridley turtles and 32% for Kemp's ridley turtles). We note that the reported conversion factors for both ridley species were from single studies with small sample sizes, so these results should be treated with due caution.

Total estimated annual turtle mortality in longline fishing under ICCAT auspices, for the 13 RMUs for which estimates could be made, amounted to ~25,000 turtles killed per annum according to the approach and stratifications we adopted here. However, there is little confidence in this figure due to the application of a single BPUE estimate, usually from a range of reported BPUEs, to large aggregations of longline fishing effort data. The values should not be used as firm estimates or taken out of context; their primary value lies in the relative impacts that are likely to accrue to each RMU. Loggerhead turtles constitute almost half of that total (>9,000). These results illustrate the different susceptibilities of species to being caught in longlines. Loggerhead turtles are carnivorous and are likely to be attracted to baited hooks. Both these and leatherback turtles have relatively large populations compared to other turtle species in the Atlantic Ocean, have trans-Atlantic crossings following the Gulf Stream, and are therefore extremely likely to interact with high seas longline fisheries. Hawksbill, ridley and green turtles have more coastal distributions, so although they may overlap to some degree with 5°x5° squares in which longline effort occurs, they probably overlap considerably less than the data available for this study suggests. On the other hand, their predominantly near-shore distributions means they are more likely to be impacted by coastal net fisheries, which are not covered in this assessment.

3.2.2 Purse seining

Clermont et al. (2012) reported a total capture of 415 turtles in >9,000 observed purse seine sets between 1995-2010 (representing 10% observer coverage). All six species occurring in the Atlantic Ocean were recorded as captured. Of these captures, 21 (5%) were reported as mortalities with another 18 (4%) where the fate was unknown, with no meaningful differences between sets on FADs versus free-schools (**Table 6**). They then scaled the BPUE to reported effort to estimate an annual bycatch of 218 turtles in the Atlantic Ocean from 1995 to 2010. Considering the low observed total bycatch (all species) and mortality we scored all but four RMUs Low

(1) as impacts are likely negligible. However, because of uncertainties in RMU sizes, assigning captured turtles to overlapping RMUs, etc. we took a precautionary approach for four RMUs (Dc-AtSW, Dc-InSW, Ei-AtE and Lo-AtE) and raised their scores to 2. It is also worth noting that although Kemp's ridley turtles constituted a significant proportion of the Atlantic Ocean purse seine catch in the Clermont et al. (2012) dataset, the authors noted that the observed effort did not overlap with the known range of Kemp's ridley turtles. The most likely explanation is misidentification of the two very similar-looking *Lepidochelys* turtle species, with most, if not all, being olive ridely turtles, not Kemp's ridley turtles.

3.2.3 Vulnerability

A plot of the all 22 RMUs impacted by longlining (**Figure 7**) combining the productivity and susceptibility scores gives a visual interpretation of how each RMUs productivity and susceptibility parameters are related. The RMUs in the top left hand corner are the most vulnerable while those towards the bottom right hand corner are the least vulnerable. The RMUs with the highest susceptibility and lowest productivity scores are the olive ridley turtles from the east Atlantic, green turtles from south Caribbean and Mediterranean loggerhead turtles. These rankings reflect the high levels of longline effort in the central tropical Atlantic Ocean and the fact that the olive ridley turtle population is one of the smallest and most data-deficient RMUs. The least susceptible RMUs are the loggerhead turtles in the Atlantic northwest, and the green and leatherback RMUs of the Indian southwest ocean. Despite the small population size of the latter RMU, overall it has a high productivity which contributes to its low vulnerability. Both the predominantly Indian Ocean RMUs are exposed to relatively low levels of longline effort where they occur in the Atlantic Ocean.

Vulnerability scores for longline and purse seine were calculated and ranked **(Table 7)**. Three of the six RMUs most vulnerable to longline fishing were also ranked in the top six for purse seine. The olive ridley turtles from east Atlantic ranked most vulnerable to both gear types; this result is driven primarily by two factors, namely conservative scoring of data-deficient productivity parameters and complete overlap with the highest levels of longline and purse seine effort. This combines with the high ranking in both vulnerability scores for both the green turtles of the south Caribbean the leatherback turtles from the Atlantic south west, to highlight the combined effects of high longlining and purse seine effort levels in the tropical Atlantic Ocean (**Figures 8-10**). There is also a smaller 'effort hotspot' in the southern Caribbean, off Venezuela. Clearly, these two areas are of highest concern for reducing impacts on turtle species. The remaining two RMUs at high risk from longlining are the loggerhead turtles from the Mediterranean and Kemp's ridley turtles from the Atlantic northwest. The relative impacts of longlining and purse seining are difficult to discriminate in the vulnerability score tables, except through inspection of the mean scores for each analysis (**Table 7**). Although the mean vulnerability score for longlining was almost 20% higher than for purse seining, the difference in likely impacts is made more evident through an inspection of the mean scores for the highest-ranked RMUs, which was some 40% higher for longlining (mean = 2.26) than for purse seining (mean = 1.39). This relative assessment is, however, probably less powerful than comparing directly the estimated annual mortality from these two gear types, with purse seine fishing accounting for trivial levels of mortality compared to longlining.

Considering species-level impacts, five of the six turtle species in the ICCAT region were recorded as being at highest vulnerability for both longline and purse seine PSAs. Thus there do not appear to be species-specific characteristics driving vulnerability.

3.3 Gap analysis

This ecological risk assessment was limited by the quality and availability of data, such that only a semi-quantitative level 2 analysis was possible. Even within this broad categorisation, many more factors that are believed to inform risk, encounter probabilities (or catchability) and survival probability after encounter, could not be quantified at all. This therefore represents a first step of what is likely to be an iterative ERA process. A recent paper by Lewison at al. (2013) reviewed the current state of knowledge in sea turtle bycatch presenting new ways forward for bycatch research and management. Here we detail data gaps relating to turtle productivity and susceptibility parameters, as well as fishing operational data and reporting, needed to improve future risk assessment for ICCAT fisheries.

3.3.1 Productivity parameters

Of the original parameters proposed for inclusion, two (generation length and numbers of adult males) were excluded because of a lack of adequate/robust data. However, because the assessment was conducted at the level of the RMU, there were data gaps for virtually all RMUs. Furthermore, many RMUs have a huge number of rookeries, ranging across biogeographic regions, yet for many population productivity parameters we had to rely on a single study from a single site. The biggest data gaps relate to estimates of annual survival for different age/stage classes, for which no data exist. This gap effectively prevents any numerical (as opposed to scenario-based) assessment of population demography and population dynamics. Without these, it is impossible to scale the impacts of fisheries mortality to natural mortality.

Although spatio-temporal distribution patterns of turtles relates more to susceptibility to fishing or encounter probabilities with fishing gear, it is an area of research for which turtle biologists and conservationists should continue to focus efforts. The RMU boundaries used here are unlikely to be the true boundaries. More data, and better data availability, will reveal areas of high or low use by different species/RMUs that would facilitate far more sensitive and believable estimates of annual fishing effort encountered by RMUs.

3.3.2 Susceptibility

Assessing susceptibility of turtles to fishing effort requires paired information – for each fishing effort stratum or parameter, a corresponding datum relating to turtle interactions is required. The data requirements increase rapidly with each additional stratum or parameter. For example, temporal patterns in risk to turtles from longline fishing could be divided by decade, by year, or by season (either summer/winter or quarter-year). While ICCAT catch and effort statistical databases contain data sufficient to create these strata, little is achieved if, in slicing the effort data in this way one merely creates data gaps because new layers have no corresponding turtle interaction data, or one is forced to assume that interaction data from one stratum can be applied to another. Some data were available to conduct temporal assessments of bycatch risk; these were insufficient to merit inclusion in the full PSA, but case studies of what is possible are shown in Box 1 for illustrative purposes.

Assessing vertical overlap requires that the actual setting depth be recorded, and can be related to the probabilities that turtles are encountered at the same depths. Self-evidently any air-breathing, diving animal will have a vertical distribution in the water column that is a mix of time spent at the surface, time spent diving down to or surfacing from target/foraging depth, and time spent at target/foraging depths (which will vary considerably). What those depths are (mean, range, etc.) and how they vary by species, season, oceanographic conditions or other parameters is poorly known. How these turtle foraging ecology parameters interact with lines that straddle these categories, and how the catena on longlines increases or decreases encounter probabilities within a single set, or how long hooks take to reach target depths or be retrieved and are brought from depth into the zone where turtles are typically, found remain unknown. Equally, gear configurations and target depths vary tremendously between and within fleets, and data on 'target species' are not recorded through ICCAT or housed in ICCAT's databases, so assigning risk profiles to effort stratified by target species was not possible.

ICCAT observer programmes should routinely record the bait and hook types on which a turtle was hooked, if it was hooked in the mouth, foul-hooked or entangled with the line, and if hooked though biting or swallowing the hook, then where/how deep. Several authors have lamented the lack of data on post-release mortality (Hays 2003, Chaloupka et al. 2004, Swimmer et al. 2006) and this remains a critical data gap. Detailed information on operational aspects (preferably recorded through mandatory logbook schemes and reported to ICCAT) will also improve our ability to model fishery impacts on non-target species; metrics for parameters such as longline soak time, target fishing depth and setting/hauling data are obvious gaps.

Another key gap is understanding data quality. Self-reported and voluntary data sources on bycatch numbers/rates are inherently less reliable than data from independent, scientific observer programmes. Furthermore, reporting turtle interactions without reference to the observed effort from which those interactions were drawn is of little value.

To undertake a more robust level 2 ERA, or to attempt a level 3 ERA for some species/RMUs would require turtle bycatch data linked to spatio-temporal effort observed, with corresponding parameters such as bait type, hook type, actual mean hook depth, start and end of fishing (soak time), soak period (day, night) and visibility of baited hooks for nocturnal sets (whether light sticks are used and cloud cover relative to moon phase). Further, turtle species identification, or RMU identity, as well as sex and size-class information of captured turtles is required, preferably based on a simple, reliable and robust measurements that observers can take reliably. In addition, turtle biology and spatio-temporal distribution patterns, by sex/age class, requires appreciably more information.

BOX 1 - Case studies of temporal stratification of bycatch and fishing effort data

The following two examples illustrate how more detailed bycatch data, such as those collected by observer programmes in the USA, Brazil and Uruguay can give a more detailed and accurate picture of fishing impacts on turtles in the ICCAT region. BPUE was calculated as the sum of all observed turtles caught by quarter, divided by the sum of the observed effort for that quarter, Below are two examples of seasonality in catch rates for turtles from the northwest and from the southwest Atlantic Ocean. We highlight the differences between these regions for loggerhead and leatherback turtles.

3.4
Case 1 - Brazil and Uruguay longline pelagic fleets

The observer programmes in Brazil and Uruguay have collected detail information on effort, primarily for longline and turtle bycatch. Domingo et al. (2006) present tabulated data of all observed longline effort and turtle bycatch by species, from 1998 to 2004, grouped by month. We grouped these data into quarters and calculated the BPUE for four species of turtles. Table B1 below shows the aggregated data and highlights periods of highest BPUE for loggerhead and leatherback turtles.

Table B1. Seasonal turtle bycatch from longline fisheries in Brazil and Uruguay (combined).

Southwest Atlantic	**Quarter 1 Jan - Mar**		**Quarter 2 Apr – Jun**		**Quarter 3 Jul - Sept**		**Quarter 4 Oct - Dec**		**Total years**	
	Tot. effort	**BPUE**	**Tot. effort**	**BPUE**	**Tot. effort**	**BPUE**	**Tot. effort**	**BPUE**	**Tot. effort**	**BPUE**
Species	422,961		453,859		1,240,600		1,032,218		**3,149,638**	
Caretta caretta	156	0.3688	491	**1.0818**	282	0.2273	199	0.1928	1128	0.3581
Dermochelys coreacea	22	0.0520	52	**0.1146**	158	**0.1274**	27	0.0262	259	0.0822
Chelonia mydas	2	0.0047	1	0.0022	19	0.0153	6	0.0058	28	0.0089
Lepidochelys olivacea	0	0.0000	0	0.0000	22	0.0177	7	0.0068	29	0.0092
Unknown	12	0.0284	0	0.0000	19	0.0153	11	0.0107	42	0.0133
Total bycath all sp.		0.4539		**1.1986**		0.4030		0.2422		0.4718

Summary results: Table B1 shows that the highest BPUE for all species is in the second quarter (BPUE=1.1986), also the quarter of highest loggerhead and leatherback turtle catches. The lowest BPUE for loggerhead turtles occurred the in fourth quarter. The leatherback turtles are mostly caught during the second and third quarter, BPUE=0.1146 and 0.1274 respectively, with the lowest BPUE in the fourth quarter (0.0262). Giffoni et al. (2012) analysed 13 years of turtle bycatch data from observer programmes in the Uruguayan and Brazilian pelagic longline fleets. They stratified the data by year and fishing areas, used by the fleet (Latitudes 19˚S and 44̊ S) arriving at similar results for both species as are described here.

Case 2 - U.S.A. North Atlantic pelagic longline fleet

The observer programme for the U.S.A. pelagic longline fleet has been in place since 1992. It documents interaction rates between the fishery and non-target species, including sea turtles. A mandatory fisheries logbook system (FLS) has also been in place since 1992, wherein records of fishing effort, gear characteristics and commercial catch are captured. Table B2 shows results of seasonal turtle catches using observer and longline fishing effort (EffDIS) datasets provided by ICCAT, from 1999 to 2009. The effort and bycatch data were stratified by year, month and species. Additional information such hook type and fate of turtles was also available; only hook type data are presented. The data were grouped to allow for a seasonal comparison between the ocean regions (see Appendix 4 for full dataset and calculations). A change in gear from J hooks to circle hooks occurred in 2005. The original data includes some captures which were categorised as both circle and j hook captures, and some were not categorised; these are excluded here. Comparisons between hook type and bycatch are not discussed, and shown just for seasonal comparisons.

Table B2. Seasonal turtle bycatch from the U.S.A. pelagic longline fleet, 1999-2009

Northwest Atlantic	**Quarter 1 Jan - Mar**			**Quarter 2 Apr - Jun**			**Quarter 3 Jul - Sept**			**Quarter 4 Oct - Dec**			**Total years**
	Hook style			Hook style			Hook style			Hook style			
	Circle	J	Total	Circle	J	Total	Circle	J	Total	Circle	J	Total	
Caretta caretta	20	51	71	29	58	87	170	56	226	44	29	73	457
BPUE	0.0013	**0.0034**	0.0048	0.0015	0.0029	0.0044	**0.0073**	0.0024	**0.0096**	0.0026	0.0017	0.0043	0.0061
Dermochelys coreacea	28	47	75	139	73	212	75	71	146	73	23	96	529
BPUE	0.0019	0.0031	0.0050	**0.0070**	**0.0037**	**0.0107**	0.0032	0.0030	0.0062	0.0043	0.0014	0.0057	0.0070
Lepidochelys kempii							1		1				1
BPUE							0.0000		0.0000				0.0000
Lepidochelys olivacea				1		1							1
BPUE				0.0001		0.0001							0.0000
Grand Total	48	99	147	169	136	305	246	127	373	117	54	171	996
BPUE (hooks/1000)	0.0032	0.0066	0.0098	0.0085	0.0068	0.0154	0.0105	0.0054	**0.0159**	0.0069	0.0032	0.0101	0.0133

Summary results: Table B2 shows BPUE for four species of turtles by hook type and quarter. The highest turtle BPUE for all species, irrespective of hook type, occurred in the second and third quarters (BPUE~0.015) with the lowest BPUE (0.001) in the first quarter. Olive and Kemp's ridley turtles where only caught during the second quarter and in the third quarter, respectively. Loggerhead turtles where mostly caught in the third quarter with circle hooks (BPUE =0.0073), and in the first quarter with J hooks (BPUE=0.0034). The highest loggerhead turtle BPUE, irrespective of hook type, was during the third quarter (BPUE=0.0096). The highest leatherback turtle bycatch was during the second quarter for both for circle and J hooks (BPUE=0.0070 and 0.0037, respectively).

Comparison between data sets

Loggerhead turtles in the southwest Atlantic are primarily caught during the second quarter compared to the northwest Atlantic where the majority of the bycatch (on circle hooks) occurs in the third quarter. The leatherback turtles are primarily caught during the second and third quarters in the southwest Atlantic and in the northwest Atlantic, which demonstrates that seasons are not necessarily good predictors for species-specific bycatch risks. Differences in migration periods, breeding seasons or foraging habits probably explain these patterns. This highlights the tremendous spatio-temporal variability in risk within and between species which, when combined with seasonal patterns in fishing effort, gear configurations, etc. can produce more nuanced assessments of risks or total mortality. However, to achieve these more fine-scaled analyses, detailed data collection (bycatch and fisheries information), and collaboration between observer programs, such as between Brazil and Uruguay, is required

4. Discussion

Fishing effort in ICCAT is dominated by two major gear types, reflected in proportional catches by gear type: purse seine (36%) and longline (33%). These two capture techniques have appreciable non-target capture risks (Wallace et al. 2010b). Further, an analysis of productivity and susceptibility for all bycatch species in the ICCAT convention area revealed that longline fisheries have the highest bycatch species diversity, followed by gillnets and purse seines (Arrizabalaga et al. 2011). Regional (intra-ICCAT) assessments and estimates of turtle bycatch from longlining have been conducted, including for the North American fleet operating in the North Atlantic (e.g. Levison & Crowder 2007, Gardner et al. 2008, Finkbeiner et al. 2011); the Brazilian and Uruguayan longline fleets in the Southwest Atlantic (e.g. Giffoni et al. 2012, Pons et al. 2012); and the Mediterranean fleets (e.g. Alessandro and Antonello 2010). Further, Clermont et al. (2012) reported on a large-scale, multi-year programme assessing interactions between turtles and purse seine fishing in the ICCAT region. The results presented here are the first attempt at assessing the relative impact of the two major ICCAT fisheries on sea turtle populations across the entire Convention Area.

A key outcome is that the direct impacts on turtles from purse seine fishing operations appears to be minor in comparison to the impacts from longline fishing. However, the overlapping areas of highly concentrated purse seine and longline effort in the tropical Atlantic Ocean combine to make the low-latitude Atlantic Ocean an area of high risk to turtles. Although significant areas of uncertainty remain, including turtle mortalities on lost FADs, and post-release mortality rate, it is unlikely that improved understanding of these will change the relative impacts by gear type or relative rankings of RMUs.

At the species level, the distribution of scores through the ranked tables is largely similar for productivity, longline vulnerability and purse seine vulnerability scores. However, productivity scores were more noticeably grouped by species than either vulnerability scores (compare **Tables 5 & 7**). Of particular interest are the management implications from a lack of species-specific patterns in vulnerability. All turtle species appear vulnerable to being captured in both longline and purse seine gear, and management measures to mitigate capture will likely benefit all species occurring in the regions in which those mitigation measures are implemented.

Turtle bycatch is dependent on many factors. Some are intrinsic to turtle biology such as foraging ecology and behavioural ecology, such as migration (Lewison et. al. 2013), while others relate directly to the method (e.g. type of gear) and strategy of fishing, which vary depending on the target species, oceanographic conditions (e.g. sea temperatures or depth of thermoclines), and fishing traditions and how those intersect with turtle ecology (Schofield et al. 2010). Pelagic longline fishing methods are very diverse but, in the context of turtle interactions, can be broadly divided into shallow vs deep sets depending on the target species. Turtle interactions have been shown to be approximately 10 times greater in longline shallow sets versus deep sets (Beverley et al. 2004, 2009). This is because turtles spend the majority of their time in the upper 100 m of the water column (Eckert et al. 1989; Polovina et al. 2003, 2004; Parker et al., 2005; Swimmer et al. 2006). Thus turtles are exposed to capture potentially for the entire soak time of a shallow longline set, whereas for deep sets they are at risk as the lines sink through the water column during deployment, and possibly again when gear is brought to the surface when it is retrieved (Watson et al. 2005). When targeting swordfish, longliners generally set between sunset and sunrise and are relatively shallow (20 m – 30 m), because swordfish have nocturnal, near-surface feeding habits (Berkeley et al. 1981). When targeting tunas, lines are typically set in the morning and hauled in the evening, with hooks set at greater depths (between 10 – >500 m). In addition, tuna-directed sets use bait only, whereas swordfish-directed sets use a combination of bait and light sticks. Light sticks have been shown to attract turtles (Witzell 1999, Wang et al. 2007). Leatherback turtles in particular are more often caught in nocturnal longline sets than in comparable diurnal sets (Gilman et al. 2006).

The types of hooks used varies between fleets/target species. Broadly speaking there are three kinds of hooks used in pelagic longline fishing: Japanese-style (offset) tuna hooks, circle hooks, and J hooks (typically used when targeting swordfish (Beverly 2006)). J hooks are associated with higher turtle bycatch rates than the other types of hooks and switching from J hooks to circle hooks tends to decrease the severity of hooking. Also the use of larger hooks (> 51 mm) has been shown to reduce the chances of turtles, in particular loggerheads, from ingesting them (Watson et al. 2005, Beverley 2006). These changes have been implemented in several fisheries in particular because the they have resulted in little impact on catch rates (Watson et al. 2005, Gilman et al. 2006, Read 2007, Pacheco et al. 2011, Swimmer et al. 2011).

This semi-quantitative, Level 2 ERA has demonstrated first that purse seine fishing poses negligible threats to turtles relative to longline fishing. Second, there are significant losses for several turtle populations from ICCAT longline fishing. Those losses cannot be fully quantified through this assessment, and there is no attempt to apportion any possible extinction risk to mortality from fishing. However there is a *prima facie* case for ICCAT CPCs to engage in more data collection on turtle interactions with their fishing operations. To be most useful, set-by-set turtle bycatch data and associated fishing operational data should be shared, so that more refined, in-depth and detailed analyses of risk can be conducted at the scale of the ocean basin. Such analyses should be geared to account for risk factors more explicitly than was possible here, including seasonality of effort and bycatch rates and risks from deep versus shallow sets. A more detailed Level 2 analysis, or a Level 3 analysis for certain species, are likely required to assess the true likelihood that longline fishing is or is not a significant driver of extinction risk to certain turtle populations in the Atlantic Ocean. Equally, more fine-scale analyses of turtles RMUs' (or other population delimitations) at-sea spatio-temporal distributions and densities (and therefore their exposure to longline fishing effort) are urgently required. While many of the datasets for assessing at-sea patterns do exist, they are not readily available for analysis such as this.

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Table 1. IUCN Red list threat status for all marine turtle species

Common name	*Species name*	*IUCN status*
Leatherback turtle	*Dermochelys coriacea*	Critically Endangered
Hawksbill turtle	*Eretmochelys imbricata*	Critically Endangered
Kemp's ridley turtle	*Lepidochelys kempii*	Critically Endangered
Loggerhead turtle	*Caretta caretta*	Endangered
Green turtle	*Chelonia mydas*	Endangered
Olive ridley turtle	*Lepidochelys olivacea*	Vulnerable
*Flatback turtle	*Natator depressus*	Data deficient

*The flatback turtle is the only one that does not occur in the Atlantic Ocean

Table 2. Turtle species regional management units in the Atlantic Ocean.

Common name	*RMU*	*RMU area*
Loggerhead turtle	Cc-AtNW	Atlantic northwest
	Cc-AtNE	Atlantic northeast
	Cc-Med	Mediterranean
	Cc-AtSW	Atlantic southwest
	Cc-InSW	Indian southwest
Green turtle	Cm-AtNW	Atlantic northwest
	Cm-AtSCar	Atlantic south Caribbean
	Cm-AtSCen	Atlantic south central
	Cm-AtSW	Atlantic southwest
	Cm-AtE	Atlantic east
	Cm-InSW	Indian southwest
	Cm-Med	Mediterranean
Leatherback turtle	Dc-AtNW	Atlantic northwest
	Dc-AtSE	Atlantic southeast*
	Dc-AtSW	Atlantic southwest*
	Dc-InSW	Indian southwest
Hawksbill turtle	Ei-AtWCar	Atlantic west Caribbean
	Ei-AtE	Atlantic east
	Ei-AtSW	Atlantic southwest
Olive ridley turtle	Lo-AtW	Atlantic west
	Lo-AtE	Atlantic east
Kemp's ridley turtle	Lk-AtNW	Atlantic northwest

*These two RMU areas are indistinguishable from each other and overlap completely at sea (drawn as one RMU on maps). Turtles caught in the South Atlantic area are assigned to one RMU only if caught close to the south eastern or south western coasts. Otherwise, they are assigned to both RMUs.

Table 3a. Scoring and categorization for productivity parameters used in the ICCAT ecological risk assessment for turtles. Scores were divided into half-points and five categories were used when parameters scaled across orders of magnitude.

Productivity score	Number of breeding females[1]	Population trend	Generation length	Age at maturity	Hatching success	Emergence success	Mean clutch size	Nests/ female/ season	Remigration interval
1 – 1.5 (Low)	Very small, Small	Declining/ Uncertain	DD	>30 years	<50%	<50%	<90 eggs	< 4 nests	> 4 years
2 – 2.5 (Medium)	Medium, Large	Stable	DD	16-30 years	50 – 75%	50 – 75%	90-120	4-6	4-2.6
3 (high)	Very large	Increasing	DD	<16 years	>75%	>75%	>120	>6	<2.6

[1]Note that these were scored differently for each species, see Table 3b

Table 3b. Categorisations used to assign scores for each sea turtle Regional Management Unit. Values relate to numbers of breeding females, following Wallace et al. (2010a)

Species	Very Small	Small	Medium	Large	Very Large
Dermochlelys coriacea	<10	10-100	100-500	500-1000	>1000
Eretmochelys imbricata	<10	10-100	100-500	500-1000	>1000
Caretta caretta	<100	100-1000	1000-5000	5000-10,000	>10,000
Chelonia mydas	<100	100-1000	1000-5000	5000-10,000	>10,000
Lepidochelys olivacea	<100	100-1000	1000-10,000	10,000-100,000	>100,000
Lepidochely kempii	<100	100-1000	1000-10,000	10,000-100,000	>100,000

Table 4. Scoring and categorization for susceptibility parameters used in the ICCAT ecological risk assessment for turtles.

Susceptibility score	***Overlap with ICCAT region (no. squares)***	***Confidence (no. of satellite tracks)***	***Bycatch mortality relative to breeding females (%)***
1 (Low)	<50	<5	<30
2 (Medium)	50 - 100	5 - 30	30 - 100
3 (high)	>100	>30	>100

Table 5. Productivity scores and rank for each sea turtle regional management unit (RMU) overlapping with the ICCAT region.

RMU	Productivity score	Rank
Cm-InSW	2.6	1
Ei-AtWCar	2.55	2
Cm-AtNW	2.45	3
Cc-AtNW	2.4	4
Cc-AtSW	2.3	5
Lk-AtNW	2.3	5
Dc-InSW	2.15	7
Cc-InSW	2.1	8
Cm-AtE	2.1	8
Lo-AtW	2.1	8
Ei-AtSW	2	11
Dc-AtSE	1.95	12
Cc-Med	1.75	13
Cm-AtSCar	1.7	14
Cm-AtSCen	1.7	14
Cm-AtSW	1.7	14
Cm-Med	1.7	14
Dc-AtNW	1.7	14
Lo-AtE	1.55	19
Cc-AtNE	1.5	20
Dc-AtSW	1.5	20
Ei-AtE	1.2	22

Table 6. Total number of sea turtles captured (by species and fate) from >9,000 observed purse seine sets in the Atlantic Ocean (1995-2011), taken from Clermont et al. (2012). Captures and fates are also presented by sets on Fish Aggregating Devices (FAD) and sets on free swimming-schools (FSC).

Species	Alive	Dead	Unknown	Total caught
Caretta caretta	67	3	3	73
Chelonia mydas	36		4	40
Dermochelys coriacea	60	4	3	67
Eretmochelys imbricata	12	2		14
Lepidochelys kempii	35	2	1	38
Lepidochelys olivacea	73	1	2	76
Unidentified	93	9	5	107
Total	**376**	**21**	**18**	**415**
Fate (%)	90.60%	**5,06%**	4,34%	100,00%
FAD	185	7	9	201
	92,04%	3,48%	4,48%	100,00%
FSC	191	14	9	214
	89,25%	6,54%	4,21%	100,00%

Table 7. Ranked vulnerability scores for turtle RMUs in the ICCAT region, by gear type (LL = longlining, PS – Purse seining).Note RMUs that don't overlap with any PS effort are excluded from the table

RMU	LL	Rank
Lo-AtE	2.47	1
Cm-AtSCar	2.39	2
Cc-Med	2.36	3
Dc-AtSW	2.12	4
Lk-AtNW	2.12	4
Cc-AtSW	2.12	4
Cm-AtNW	2.07	7
Ei-AtE	2.06	8
Ei-AtWCar	2.05	9
Dc-AtNW	1.98	10
Cc-AtNE	1.80	11
Cm-AtE	1.75	12
Dc-AtSE	1.45	13
Ei-AtSW	1.41	14
Cm-Med	1.39	15
Cm-AtSW	1.39	15
Cm-AtSCen	1.39	15
Lo-AtW	1.35	18
Cc-AtNW	1.17	19
Cm-InSW	1.08	20
Cc-InSW	1.03	21
Dc-InSW	0.99	22
Mean	1.72	

RMU	PS	Rank
Lo-AtE	2.09	1
Ei-AtE	1.87	2
Cc-AtNE	1.80	3
Dc-AtSW	1.80	3
Cm-AtSCar	1.64	5
Dc-AtNW	1.64	5
Dc-AtSE	1.45	7
Cm-AtSCen	1.39	8
Cm-AtSW	1.39	8
Dc-InSW	1.31	10
Cc-AtSW	1.22	11
Cc-AtNW	1.17	12
Cm-AtNW	1.14	13
Ei-AtWCar	1.10	14
Cm-AtE	1.03	15
Lo-AtW	1.03	15
Ei-AtSW	1.00	17
Mean	1.416	

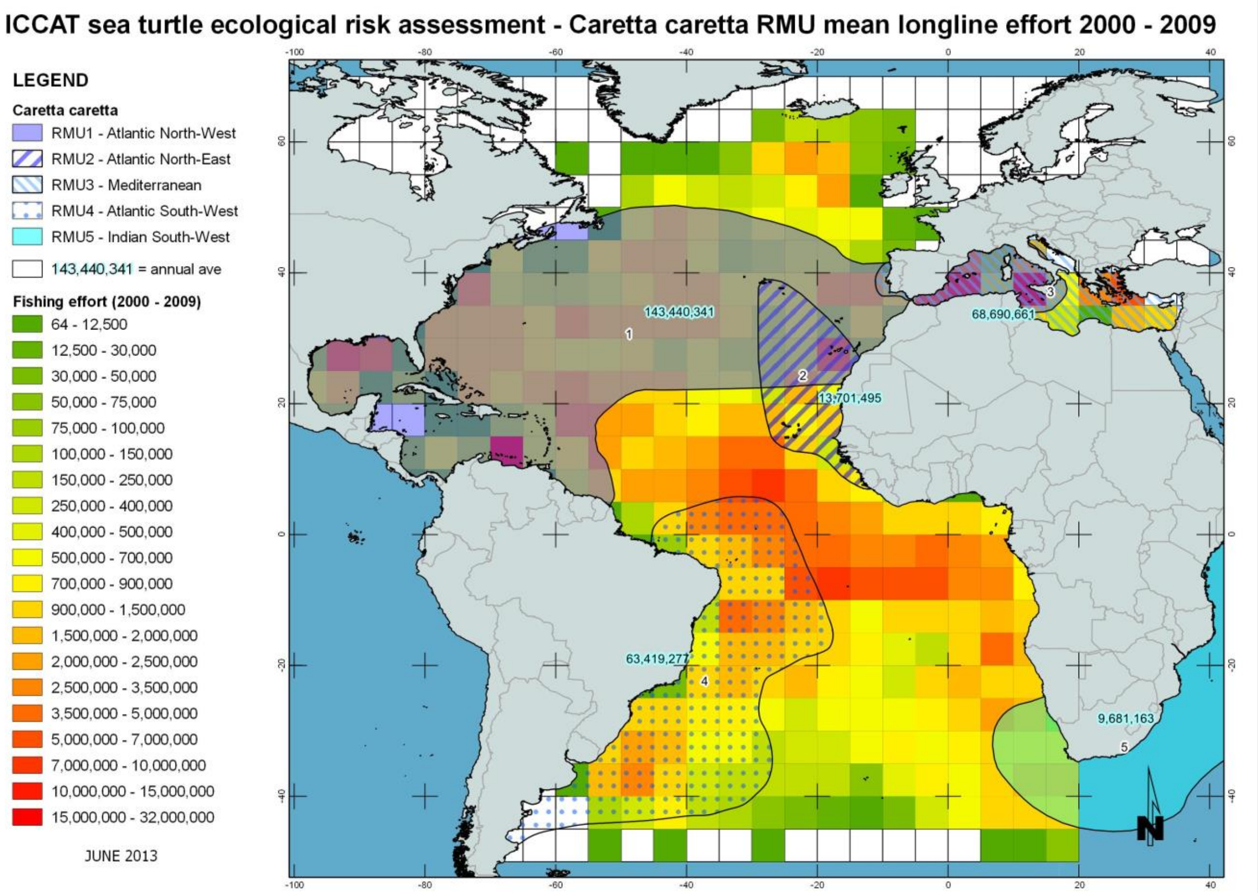

Figure 1. Mean longline fishing effort (2000-2009) in the ICCAT region overlapped with *Caretta caretta* (loggerhead) turtle species Regional Management Units (shaded polygons). The highlighted numbers show the total estimated number of hooks to which each RMU is exposed.

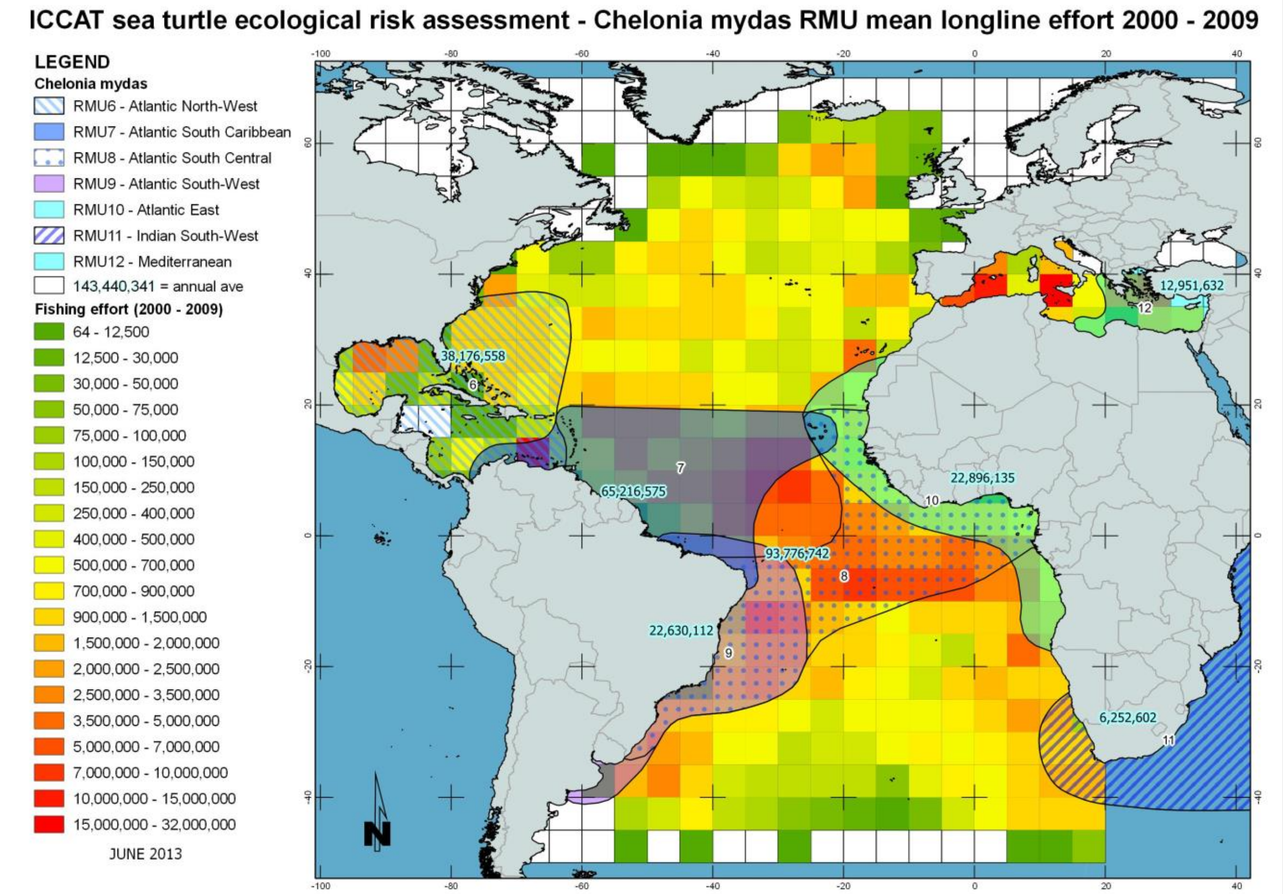

Figure 2. Mean longline fishing effort (2000-2009) in the ICCAT region overlapped with *Chelonia mydas* (green) turtle species Regional Management Units (shaded polygons). The highlighted numbers show the total estimated number of hooks to which each RMU is exposed.

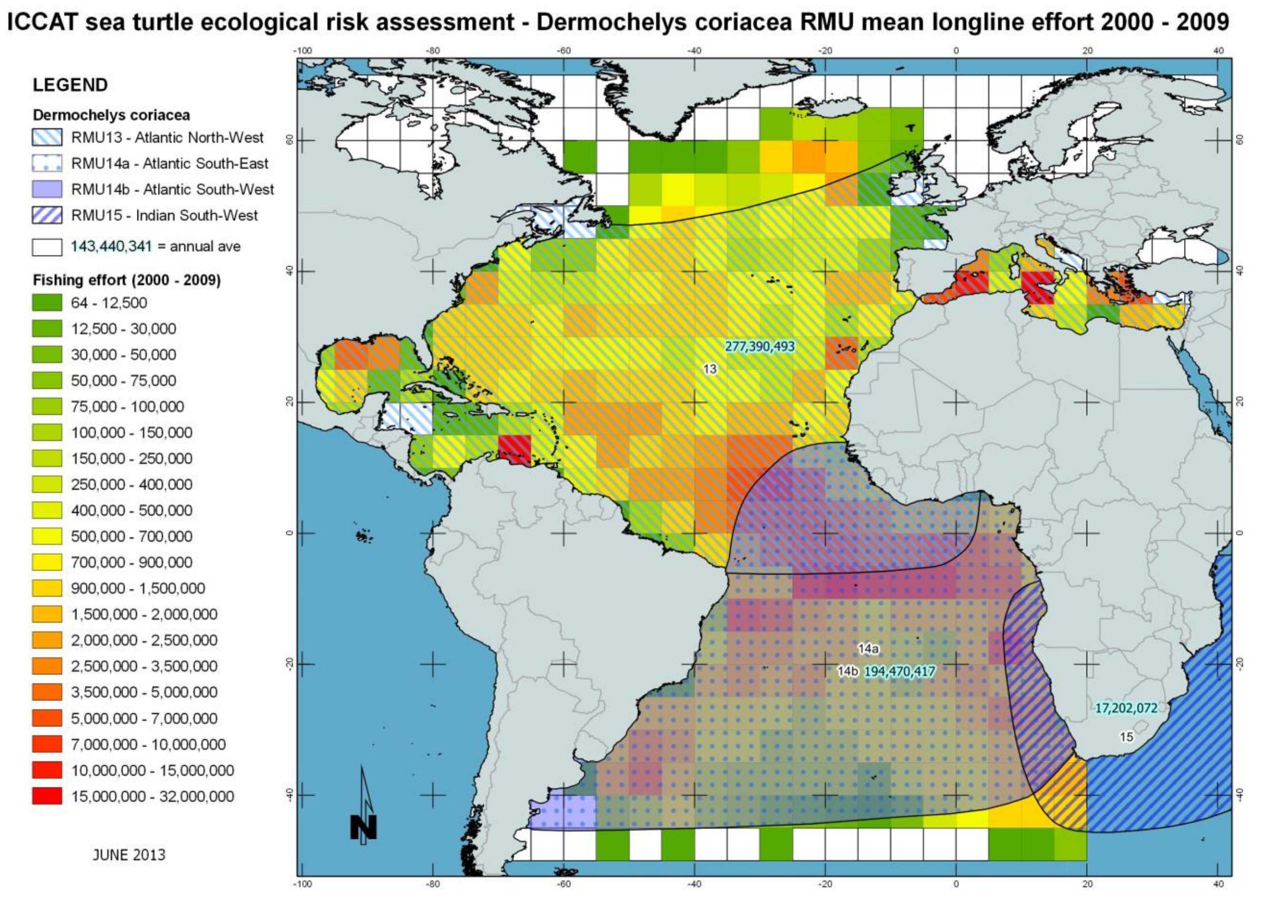

Figure 3. Mean longline fishing effort (2000-2009) in the ICCAT region overlapped with *Dermochelys coriacea* (leatherhead) turtle species Regional Management Units (shaded polygons). The highlighted numbers show the total estimated number of hooks to which each RMU is exposed.

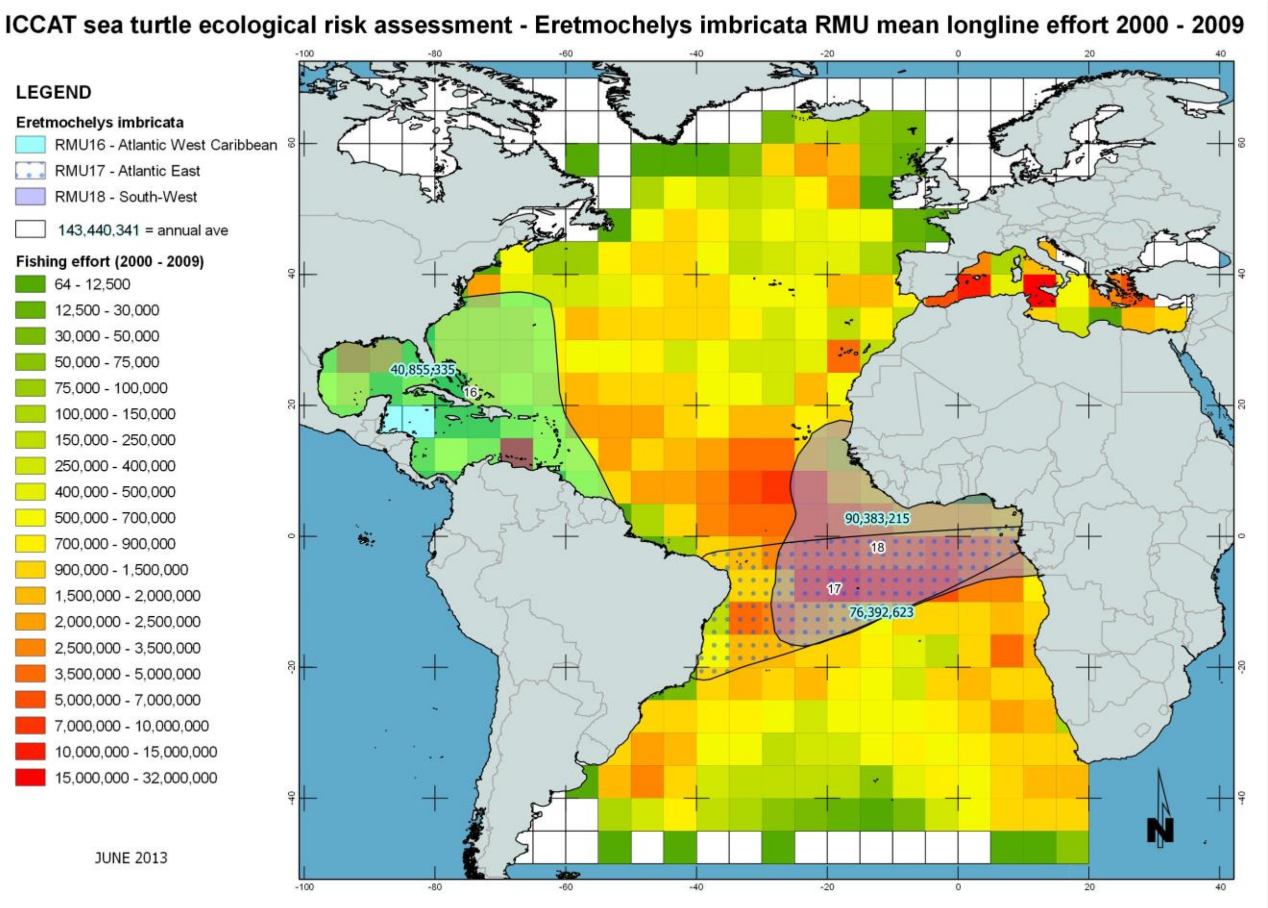

Figure 4. Mean longline fishing effort (2000-2009) in the ICCAT region overlapped with *Eretmochelys imbricata* (hawksbill) turtle species Regional Management Units (shaded polygons). The highlighted numbers show the total estimated number of hooks to which each RMU is exposed.

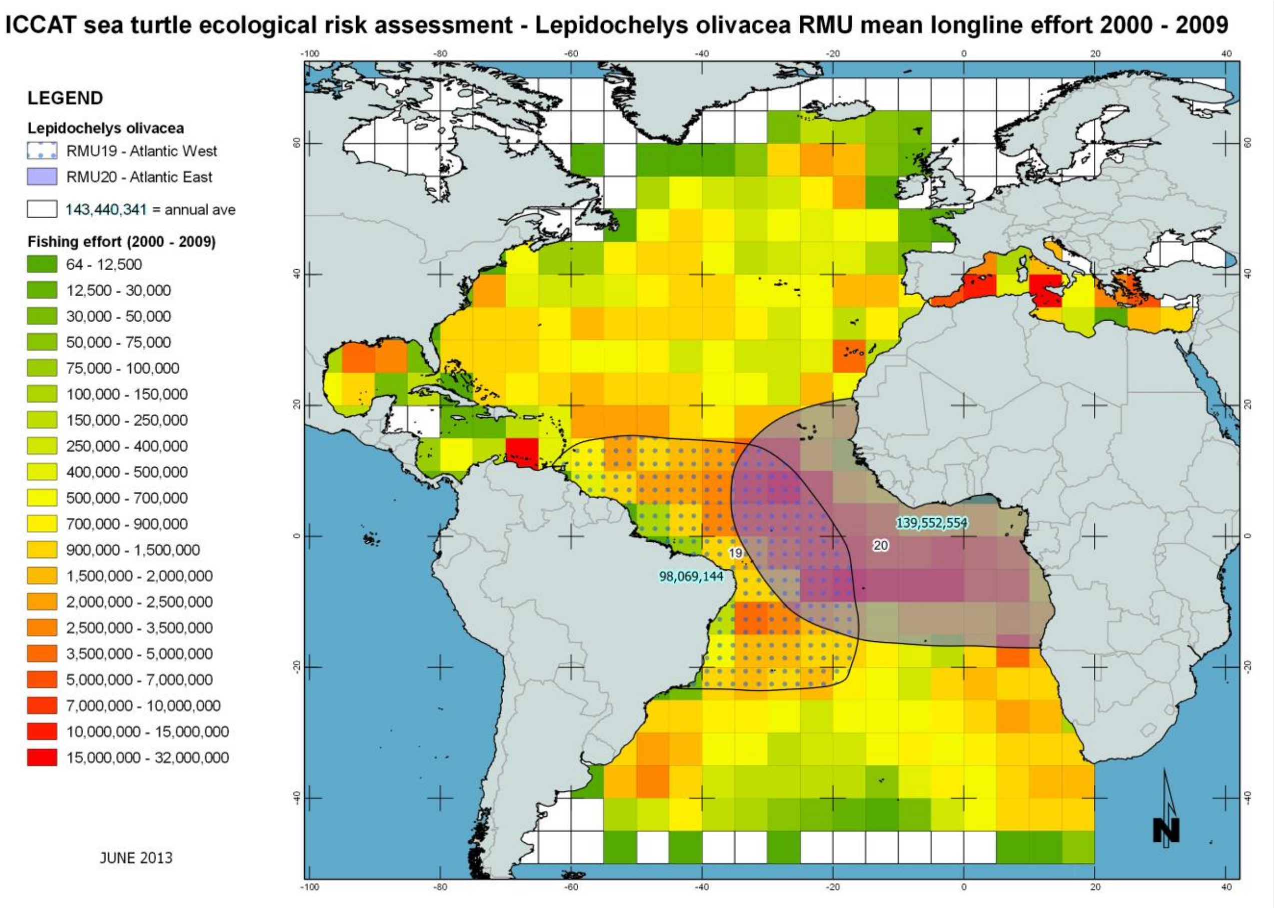

Figure 5. Mean longline fishing effort (2000-2009) in the ICCAT region overlapped with *Lepidochelys olivacea* (olive ridley) turtle species Regional Management Units (shaded polygons). The highlighted numbers show the total estimated number of hooks to which each RMU is exposed.

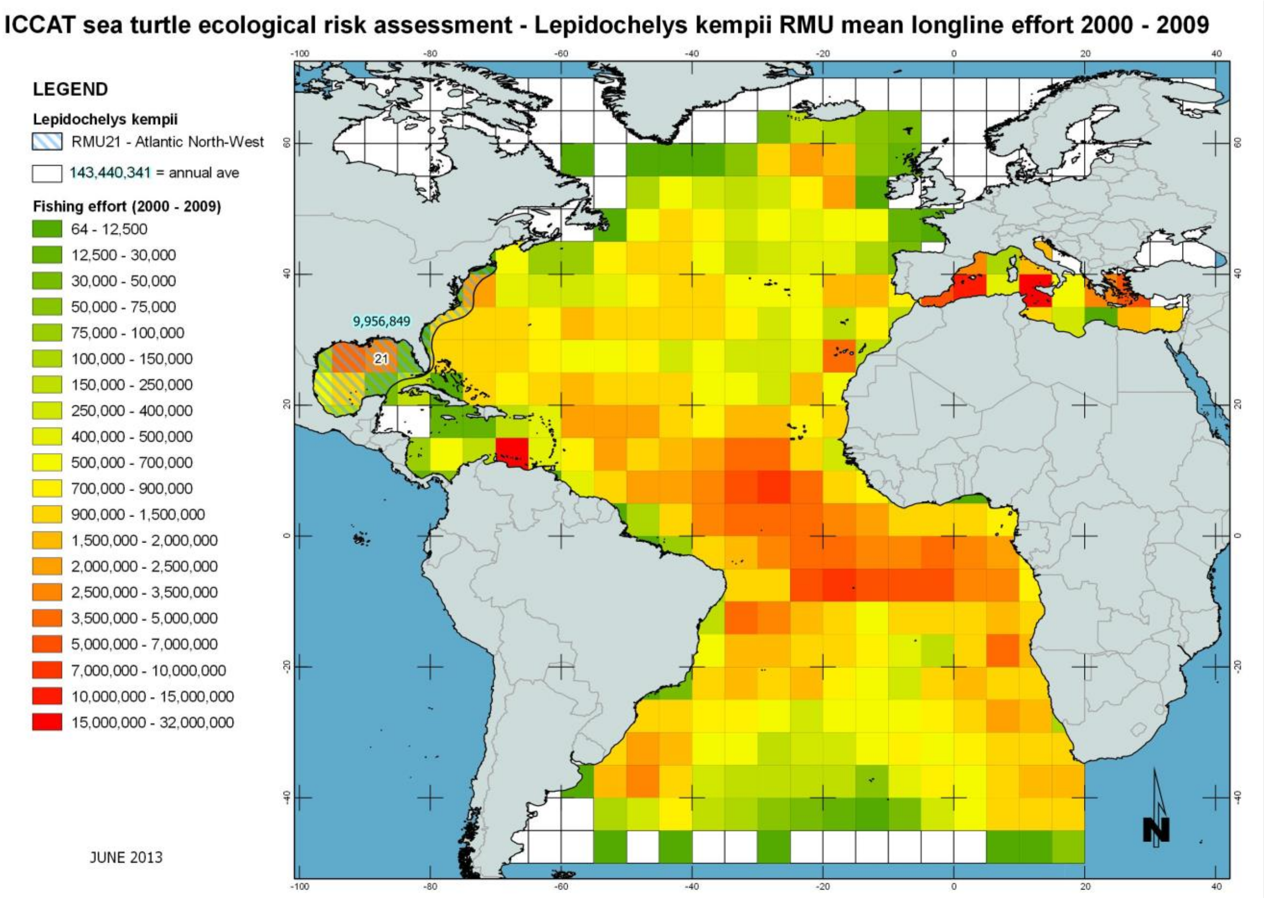

Figure 6. Mean longline fishing effort (2000-2009) in the ICCAT region overlapped with *Lepidochelys kempii* (Kemp's ridley) turtle species Regional Management Units (shaded polygons). The highlighted numbers show the total estimated number of hooks to which each RMU is exposed.

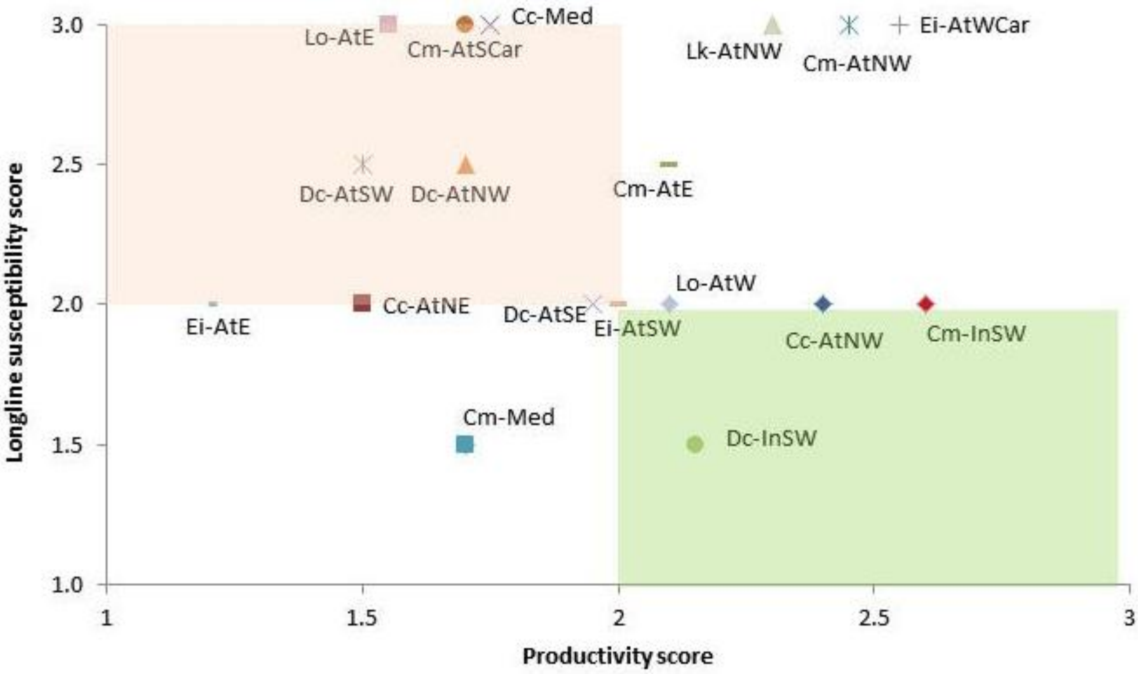

Figure 7. Productivity (x) and Susceptibility (y) scores for longline fishery in 22 sea turtle RMUs in the ICCAT region. Top-left corner = most vulnerable and bottom-right least vulnerable.

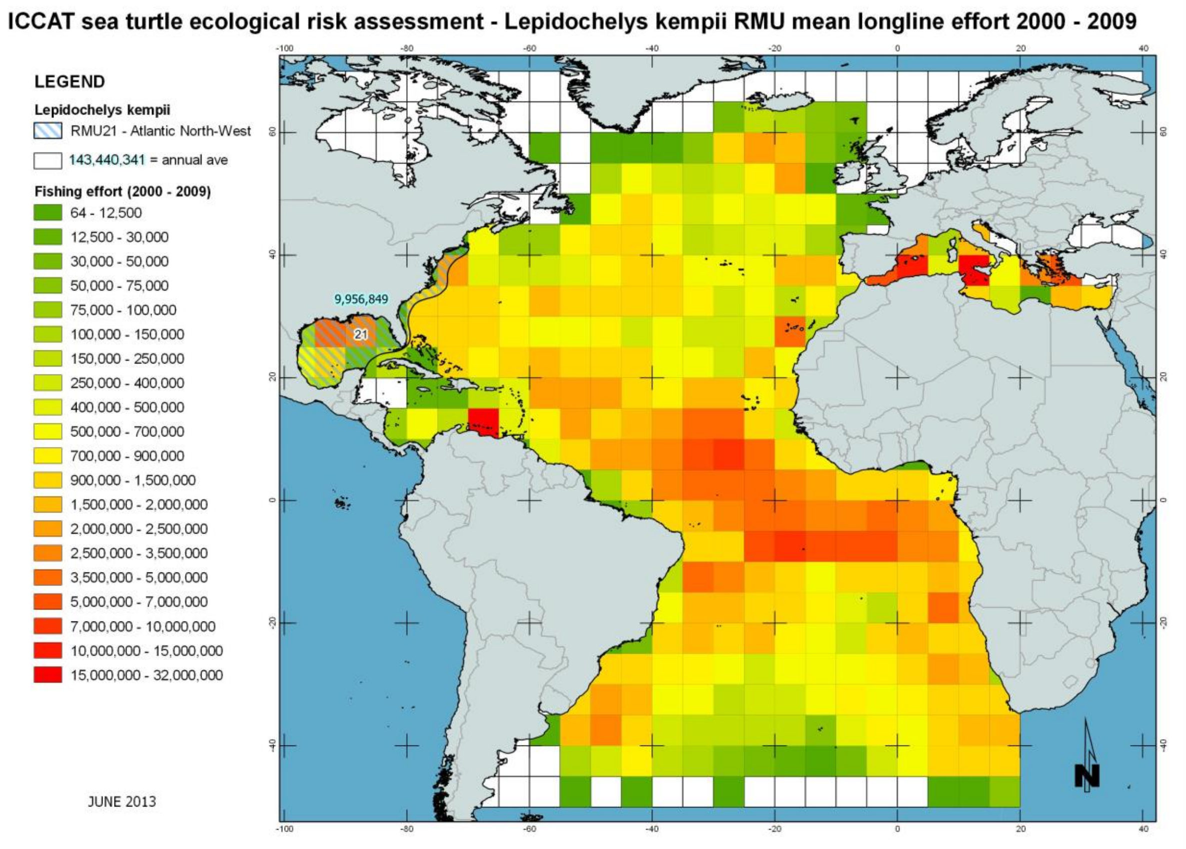

Figure 8. Mean longline fishing effort (2000-2009) in the ICCAT region, showing concentrated effort in the Mediterranean, and tropical Latitudes.

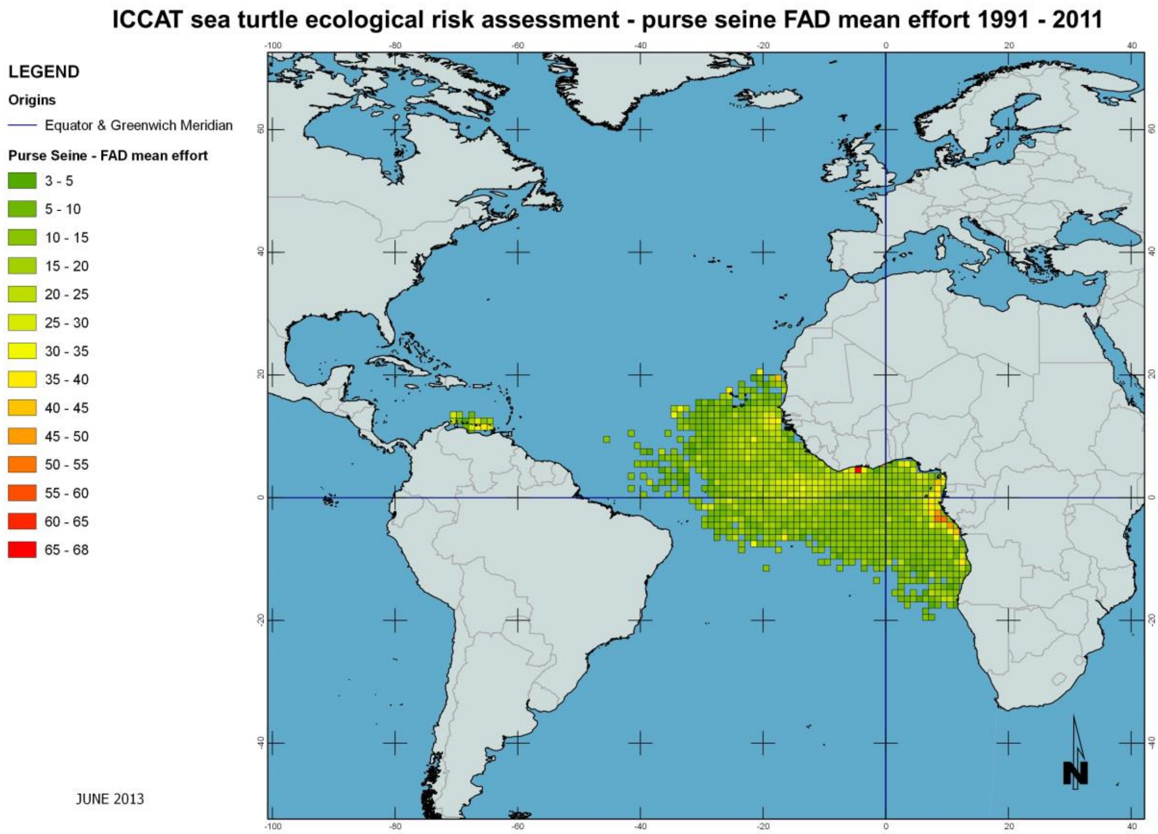

Figure 9. Mean purse seine fishing effort (1991-2011) in the ICCAT region, using fish aggregating devices (FADs). The effort is concentrated in the tropics, off West Africa between Namibia and Mauritania, and off Venezuela.

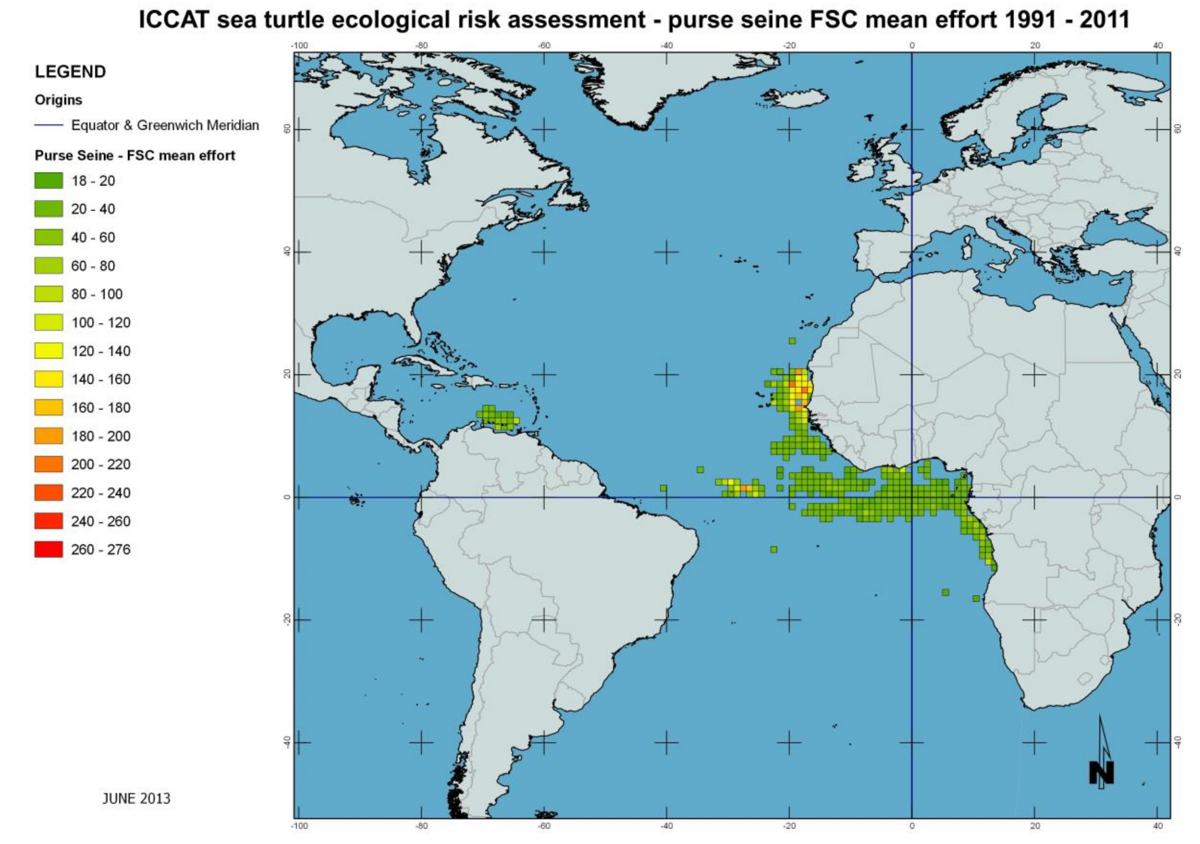

Figure 10. Mean longline purse seine effort (1991-2011) in the ICCAT region, sets over free swimming-schools (FSC). The effort is concentrated in the tropics, off West Africa between Namibia and Mauritania, and off Venezuela.