Master's thesis

NTNU
Norwegian University of Science and Technology
Faculty of Engineering
Department of Energy and Process Engineering

Marthe Alnes Høiberg

Improving the link between entanglement rates and geographic areas of plastic debris exposure

Master's thesis in Industrial Ecology
Supervisor: Francesca Verones
July 2020

NTNU
Norwegian University of
Science and Technology

Abstract

Plastic debris is found throughout the marine environment, from the sea surface to the sediments. Marine fauna has been known to interact with and become entangled in this durable anthropogenic material for decades. Despite awareness about the biodiversity threat it constitutes, quantified impacts of mismanaged plastic waste on marine ecosystems have largely remained absent from environmental assessments. In an attempt to fill this methodological gap, efforts to develop an effect factor (EF) accounting for macroplastic debris entanglement have been made in the Life cycle assessment (LCA) community. However, to capture the non-uniform spatial distribution of plastic debris and species differences in sensitivities, entanglement records for populations of species need to be related to corresponding geographic areas of plastic debris concentrations on a more population-specific level. This thesis contributes to the EF modelling development, by improving the robustness of the link between entanglement rates for selected marine species and population-specific geographic areas of exposure to macroplastic debris. For this purpose, potential areas of exposure were delineated in a population-specific spatial context to identify and estimate the exposure to plastic debris, using literature data, species distribution and occurrence maps and an existing plastic debris concentration model. By coupling this with observed entanglement rates, preliminary effect concentrations (EC) for the different species were derived. The EC's were used to rank the species in the current dataset by their predicted sensitivity to macroplastic debris, and can further be used as concentration-response data in a future species sensitivity distribution (SSD) based EF model. The sensitivity of the estimated EC's was found to be influenced by the geographical extent of the delineated exposure area, which highlights the need for the coupling between entanglements rates and potential areas of exposure to be robust.

Sammendrag

Plastavfall finnes i hele det marine miljøet, fra havoverflaten til sedimentene. I flere tiår har det vært kjent at marine organismer samhandler med og blir viklet inn i dette ikke-nedbrytningsbare menneskeskapte materialet. Til tross for bevissthet rundt trusselen det utgjør for det biologiske mangfoldet har kvantifiseringer av disse virkningene i stor grad uteblitt fra miljøvurderinger- og analyser. For å tette dette metodologiske hullet har forsøk på å utvikle en effektfaktor (EF) som kan kvantifisere marine dyrs innvikling i makroplast blitt utført i livssyklusanalyse (LCA) miljøet. For å fange opp de romlige variasjonene i distribusjonen av plastavfall samt arters ulike grad av sensitivitet, er det et behov for at plastinnviklingsrater for artsbestander er relatert til tilhørende geografiske områder med konsentrasjoner av plastavfall på et mer populasjonsspesifikt nivå. Denne avhandlingen er et bidrag til utviklingen av EF-modellen gjennom å forbedre robustheten til koblingen mellom plastinnviklingsrater for et utvalg av marine arter og bestandsspesifikke geografiske områder for makroplasteksponering. For dette formålet ble potensielle eksponeringsområder avgrenset i en populasjonsspesifikk kontekst for å identifisere og estimere eksponering for plastavfall, ved bruk av litteraturdata, artsdistribusjonskart og en eksisterende modell for konsentrasjoner av plastavfall. Gjennom å koble dette med observerte plastinnviklingsrater kunne foreløpige effektkonsentrasjoner (EC) for de forskjellige artene bli avledet. Disse ble brukt til å rangere artene i datasettet etter deres modellerte sensitivitet til makroplastviklinger, og kan videre brukes som konsentrasjon-respons data i en fremtidig «species sensitivity distribution» (SSD) basert EF-modell. De ble funnet at de estimerte EF verdiene var følsomme for det geografiske omfanget av det avgrensede eksponeringsområdet, noe som videre understreker behovet for at koblingen mellom plastviklingsrater og potensielle eksponeringsområder er robust.

Acknowledgements

This master thesis was written with an aim of contributing to the ongoing work towards the inclusion of the environmental impacts of marine plastic debris in Life cycle assessments (LCA). My biggest gratitude goes to my dedicated supervisors Francesca Verones and John Woods for providing valuable guidance and insight during this process, which lead to the completion of this thesis. I would also like to thank all my fellow students in the Industrial Ecology master’s program at NTNU, as well as family and friends for the support.

Table of Contents

List of figures

List of tables

Abbreviations

EC	Effect concentration where a percentage of individuals are observed affected by a stressor
EC_5	Effect concentration where 5% of a population is affected by a stressor
EC_{50}	Median effect concentration where 50% of a population is affected by a stressor
EF	Effect factor
FF	Fate factor
GIS	Geographic information systems
IUCN	International Union for Conservation of Nature
LCA	Life cycle assessment
LCIA	Life cycle impact assessment
nmi	Nautical miles
NOAA	U. S. National Oceanic and Atmospheric Administration
SSD	Species sensitivity distribution

1 Introduction

Plastics, or synthetic organic polymers, have become an integral part of human life since mass manufacturing commenced in the 1950's (Barnes et al., 2009). The production volumes and subsequent inadequate deposition of this non-biodegradable material has resulted in an accumulation of mismanaged plastic waste in landfills and the natural environment (Geyer et al., 2017). The input of mismanaged plastic waste to the environment was estimated to be between 60-99 million tonnes in 2015 alone, a figure expected to triple by 2060 following a business as usual scenario (Lebreton and Andrady, 2019). By wind, drainage and sewage systems, plastic waste is further transported into the marine environment (Barnes et al., 2009, UN Environment, 2018), in addition to the direct inputs occurring at sea from the fishery, shipping and aquaculture industries (Ryan, 2015). As plastic is often lightweight, it can easily be transported with ocean currents over vast distances (Law, 2017), which further contributes to its global omnipresence. Depending on the interaction between environmental processes and plastic properties such as size and density (Schwarz et al., 2019), the debris is distributed throughout the water column, from the surface layer (Ter Halle et al., 2017) to the deep-sea floor (Booth et al., 2017).

Marine plastic debris exists in a spectrum of sizes from the nano- to the macroscale, which in turn enables interactions with organisms of all sizes (GESAMP, 2019). Detrimental modes of interaction with fauna include ingestion and entanglement, in addition to the potential for alien species to utilize the debris as a raft, and the indirect implications of disturbed habitats and smothered vegetation (Gregory, 2009, Kühn et al., 2015, Rochman et al., 2016). The negative effects that plastic debris inflicts on seabirds, mammals and turtles have been acknowledged for decades (Shomura and Yoshida, 1985). Particularly, the lethal potential of macroplastic entanglements of larger bodied marine animals is well established (Wilcox et al., 2016), as it represents a more readily observable effect (Gall and Thompson, 2015, Werner et al., 2016).

Not only does plastic debris entanglement entail obvious welfare implications for the individual, but it also constitutes a threat to marine biodiversity (Galgani et al., 2013). As a documented cause of mortality (Gall and Thompson, 2015), it may exacerbate a decline in populations of vulnerable species. Entanglement events currently constitute the most lethal threat to the small population of the endangered North Atlantic Right Whale (*Eubalaena glacialis)*, responsible for 85% of diagnosed mortalities since 2010 (Kraus et al., 2016). Furthermore, the prevalence of entanglement injuries and mortalities on the population level is likely understated, owing to the difficult and costly endeavor of systematically observing wide ranging marine megafauna such as mammals, turtles and seabirds (Wilcox et al., 2016, Martins et al., 2019). In recent years, there has been an increased focus on ingestion of plastic debris, but quantitative assessments of entanglements are still limited (O'Hanlon et al., 2019). In general, a standardized and operational methodology for quantifying the impacts of plastic debris on ecosystems is lacking (Boucher et al., 2019).

Life Cycle Assessment (LCA), a renowned tool for assessing environmental impacts of the full life cycle of products and industrial systems (Curran, 2008), is commonly applied to quantify trade-offs with the use of different materials, such as plastic. The lack of an operational method accounting for the impacts of mismanaged plastic on ecosystems in the assessments has sparked a focus on this matter within the LCA community (Woods et al., 2016, Sonnemann and Valdivia, 2017). The inclusion of this aspect of plastic is imperative for a more accurate evaluation of the consequences of our current consumption and waste generation volumes, which in turn could aid in the development of urgently needed policy responses to plastic pollution. In addressing this methodological gap, preliminary LCA effect factors (EF) for characterizing marine biodiversity loss attributed to macroplastic debris entanglement have been developed (Woods et al., 2019, McHardy, 2019). However, the challenge of matching the spatial dispersion of plastic debris with relevant species distributions of observed entangled populations has been outlined (Woods et al., 2019). In addition to the non-uniform distribution of different types of plastic across oceanic regions (Barnes et al., 2009), species specific ecology and behavior further influence a population's exposure and sensitivity to plastic debris (Werner et al., 2016).

While existing species distribution maps collated by e.g. the International Union for Conservation of Nature (IUCN, 2020) can be applied to understand the global spatial pattern of species, they have limited utility in terms of predicting the core areas that regional populations utilize, and thus where these populations are likely to get entangled in debris. By applying general species ranges, the relationship between plastic concentrations and prevalence of entanglement incidents cannot be comprehended, which hampers the development of models for quantifying the effects of marine plastic debris. Owing to this, spatially explicit population ranges are required to attain a more relevant coupling between entanglement rates obtained from specific populations and geographic areas of exposure. The coupling needs to be robust as the relation between concentration and response serves as the basis for deriving species sensitivity distribution (SSD) based effect factors (EF). Improving this link contributes to the further development of an EF, which is a prerequisite to the inclusion of a characterization factor (CF) accounting for entanglement in macroplastic debris in LCA.

This thesis is contributes to this development by building on an existing database of entanglement data (McHardy, 2019), and increasing the robustness of the link between entanglement rates of selected marine species and corresponding estimates of exposure to plastic debris. Firstly, background information about plastic debris, entanglements and species ecology is presented, in addition to relevant concepts related to the tool LCA. Following this, potential areas of exposure for species- and location-specific entanglement rate estimates are delineated. The delineations are achieved based on data collection on the ecology of the species, such as breeding and migration behaviors and how these seasonal events influence species distributions. By accounting for these factors in a population-specific context, the geographical extents of the potential exposure areas are specified to the populations of which entanglement estimates are obtained for. This is done as a step towards generating the concentration-response data to be used in an SSD-based EF model. Although it is beyond the scope of this thesis to develop the final EF, the sensitivity of the estimated effect concentrations to range choices and plastic summary statistics applied in the present work is assessed.

2 Background

2.1 Plastic debris in the marine environment

Plastic is a collective term used for a broad and versatile group of synthetic polymers made up of chain-like molecules with high average molecular weight (Law, 2017). The polymers can be manufactured into a variety of products that have different chemical composition, specific density, size, shape and structure (Hidalgo-Ruz et al., 2012). The different properties combined with hydrodynamic processes and other environmental factors in a given area influence how mismanaged plastic is dispersed once in the marine environment (Schwarz et al., 2019). Owing to this as well as differences in the regional input of plastic waste, some locations serve as hotspots for accumulation (Figure 1). For example, five main gyres, or circular ocean currents, have been identified as locations in the open ocean where plastic debris accumulate in high concentrations (Lebreton et al., 2018) due to interactions between the rotation of the earth and wind (Mendenhall, 2018). In addition, areas with large coastal populations and high levels of maritime traffic such as the Mediterranean sea (Compa et al., 2019) and the North and Baltic seas (Van Franeker et al., 2011, Unger et al., 2017) are also known to be hotspots for marine plastic pollution. Moreover, high intensity of fishing industry along coastlines creates zones where fishery-based plastic debris is abundant (Kim et al., 2014). Marine areas further away from densely populated places such as the Southern Ocean are generally reported to have the lowest concentrations of plastic debris (Barnes et al., 2009, Eriksen et al., 2014). However, ocean currents ensures that plastic debris is dispersed also to these more pristine areas (Waluda et al., 2020).

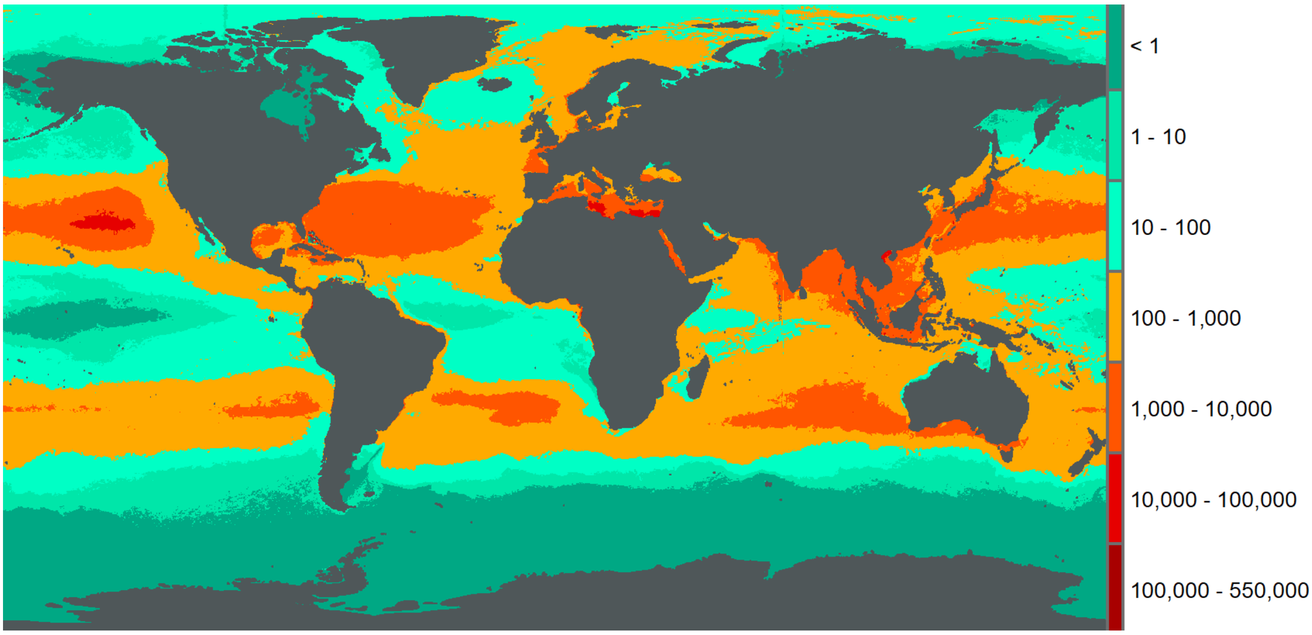


Figure 1: Global marine macroplastic (>4.75 mm) debris concentration (g/km^2). Predicted values range from 0.28 g/km^2 to a maximum value registered at 554 kg/km^2. Reproduced from data by Eriksen et al. (2014).

Plastic of different sizes are not found in equal concentrations in all compartments of the marine environment, and size can as such be a key predictor for the spatial distribution of plastic debris (Bond et al., 2018). As plastic particles and items come in a spectrum of sizes, they can be classified according to size in numerous and sometimes ambiguous ways (Hartmann et al., 2019, Filella, 2015). The simplest form is a binary distinction between microplastic (<5 mm) (Arthur et al., 2009) and macroplastic (>5 mm). Considering that the topic of this thesis relates to plastic entanglement exclusively, only macroplastic (>5 mm) is considered further, as there are currently no in situ records of microplastic entanglement (GESAMP, 2015).

A major group of macroplastic debris, which is released directly into the marine environment is that of abandoned, lost or otherwise discarded fishing gear (FAO, 2016). This includes gear such as nets, lines, ropes and various types of traps, which depending on the specific properties of the item can be found drifting on the surface, in the water column or on the sea floor (Matsuoka et al., 2005). In addition to this, packaging items constitute 40% of the total demand for plastic within Europe (PlasticsEurope, 2019), and is found as debris throughout the marine environment. Packaging items are often single-use, cheap and lightweight, which are characteristics that may facilitate careless and improper disposal (Laist, 1987) as well as allowing it to be easily moved by wind and distributed into the marine environment (Law, 2017). These two product categories, namely fishing gear and single use plastic items, constitute the most common debris items causing adverse effects on marine fauna (Rochman et al., 2016, Wilcox et al., 2016).

2.2 Marine fauna and plastic debris entanglement

Entanglement has been defined by Laist (1997) as an interaction between marine life and anthropogenic debris, that entraps animals or entangles their appendages through the loops and openings of the debris. Strapping bands, ropes or plastic bags are examples of items that may encircle or form a loop around an animal (Law, 2017), which may cause lacerations, infections and subsequent mortality (Dolman and Moore, 2017). Entrapping of animals are most commonly caused by nets and various fishery traps, an event referred to as "ghost fishing", as the abandoned gear can continue to trap and kill numerous of individuals for years after being discarded (Matsuoka et al., 2005, FAO, 2016). The type of fishing gear that animals get entangled will vary spatially depending on local fisheries activities (Raum-Suryan et al., 2009).

A difference in the abundance and distribution of fishing specific plastic debris and general plastic debris creates specific entanglement hazards in some locations (Pemberton et al., 1992). Moreover, the characteristics of the plastic debris type will influence which species are susceptible to becoming entangled in a specific item. For example, the mesh size of a discarded net will determine the size of the organisms it may trap (Sancho et al., 2003) while the buoyancy properties of the net determines its distribution in the water column (Ter Halle et al., 2016, Andrady, 2015) and as such, which species may encounter the debris. Owing to this, benthic species are more likely to be entrapped by nets or traps on the sea bottom, while larger marine mammals such as cetaceans and pinnipeds are more prone to getting entangled around their neck by various macroplastic items (Kühn et al., 2015) distributed in the upper layers of the ocean. Moreover, coastal marine animals are more likely exposed to localized sources of plastic debris, while those ranging in the open ocean may interact with plastic debris in accumulation zones (Lusher et al., 2018).

The morphology of species will also influence what type of plastic debris that may entangle an animal. It has been suggested that smaller body size can be linked to more serious injuries following the entanglement, owing to their size relative to the material (Pemberton et al., 1992). Larger mammals may be able to live longer with an entanglement, although it is likely to entail serious costs in terms of energy expenditure and mobility (van der Hoop et al., 2017).

2.2.1 Species behaviors linked to entanglement

In addition to factors related to the distribution of plastic debris, and the plastic item itself in relation to the morphology and physiology of the animal, the behavior of species will influence their exposure and sensitivity. For example, the behavior of juvenile seals is characterized by inexperience and curiosity, making them more likely to put their head through plastic items and get entangled (Werner et al., 2016, Pemberton et al., 1992). Moreover, different feeding strategies (suction, filtering, predatory and grazers) identified for species of cetaceans results in widely different habitat preferences and in turn differences in their plastic debris exposure (Lusher et al., 2018). Seabirds such as the Northern Fulmar (*Fulmarus glacialis*) and the Northern Gannet (*Morus bassanus*) feed by scavenging among floating plastic debris, and are as such prone to acquiring entanglements while foraging (Coe and Rogers, 1997). In addition, diving seabirds that pursue fish underwater are susceptible to entanglement in derelict fishing nets (Gilardi et al., 2010). Generally, foraging is assumed as the main activity of which seabirds get entangled in marine plastic debris (Thiel et al., 2018).

Foraging constitutes an integral part of the life history and ecology of animals and the foraging habits of any marine species plays a pivotal role in their specific exposure to plastic debris at sea. Owing to this, the spatial foraging patterns of marine species may also reflect where they are most likely to encounter and get entangled in plastic debris. Although foraging patterns can be difficult to ascertain as they are influenced by spatially and temporally variable environmental gradients (Votier et al., 2011, Gaston, 2004), studies that monitor and track individuals over longer periods of time can be used to quantify general patterns of the foraging ranges of populations of species (Thaxter et al., 2012).

Many seabirds and pinnipeds need to return to terrestrial sites in between foraging trips for breeding, provisioning of young and resting (Womble et al., 2009), a behavior known as central place foraging (Orians and Pearson, 1977). Owing to this, foraging ranges can be quantified as the distances they travel from the terrestrial site, i.e. the central place. For species that do not utilize terrestrial sites and are wide-ranging, obtaining population specific foraging distribution patterns is more complex. However, core areas that these species utilize can be investigated by monitoring the populations over time and identifying areas where several individuals aggregate (Waggitt et al., 2020).

2.2.2 Approaches for estimating entanglement incidence

Attempts to quantify the prevalence of entanglements within populations of species have been made using several different methods. An entanglement rate generally refers to the number of entangled individuals out of a certain sample size. While some rates are given as the annual share of entangled individuals obtained by multi-year observations of the same populations (Waluda and Staniland, 2013), others refer to a cumulative number of stranded individuals found entangled over a larger and less defined geographic area (Adimey et al., 2014). Although both types of data can be valuable in quantifying entanglement incidents, the comparability between the rates that these studies provide is low, as one represents observed chronic entanglements over a population estimate, while the other represents mortality attributed to entanglements over total number of stranded individuals. Entanglement rates obtained from stranding data is considered a cost-effective method, but the statistical credibility of this type of rate is disputed as the geographical origin of carcasses is difficult to ascertain, and the sampling is largely opportunistic in nature (Peltier et al 2012). Moreover, as the sample size is often based on total stranded individuals, the entanglement occurrence is then seen in relation to other injured and stranded individuals as opposed to the general population.

It is widely recognized that the number of stranded entangled individuals represent a minimum estimate of the prevalence of entanglement mortalities, as only a small fraction of carcasses are expected to be washed ashore and noticed (Williams et al., 2011, Panti et al., 2019, Gregory, 2009). However, for many wide-ranging mammals that are not commonly observed, stranding data provides the best available approach for investigating marine debris interactions (Unger et al., 2017). In addition, it may be argued that most other forms of observing entangled animals will inherently also undermine the prevalence of entanglement incidents. For example, entanglement rates obtained from land based surveys are also expected to be gross underestimates, as an entangled individual is less likely to return to and be spotted on land (Coe and Rogers, 1997). The carcass is in most cases more likely to disappear from the surface due to sinking, decomposition or predation (Gregory, 2009).

Applying population estimates as the sample size rather than the number of stranded individuals is a way to standardize entanglement rates across observation methods. Although it will cause underestimated rates owing to the reasons stated above, such a conversion increases the comparability across studies and populations of species, making it a more useful metric in analyses of more than one entanglement study. Although seldomly found in any studies today, the inclusion of observer effort in the rates should be an aim to further increase comparability (McIntosh et al., 2015).

2.3 Life Cycle Impact Assessment

Life Cycle Assessment (LCA) is a widely used and recognized tool applied to make holistic assessments of the environmental impacts of the entire value chain of industrial systems (Curran, 2008). This is achieved by accounting for all stages of a products life cycle, from resource extraction to waste disposal (SAIC and Curran, 2006). LCA is used to compare the environmental performance of different industrial alternatives at several levels (Jolliet et al., 2018), and can as such be used to inform policy choices. The required inputs of resources with associated emissions to the environment generated by a production system are in an LCA converted to impact scores and assigned to environmental impact categories (Figure 2).

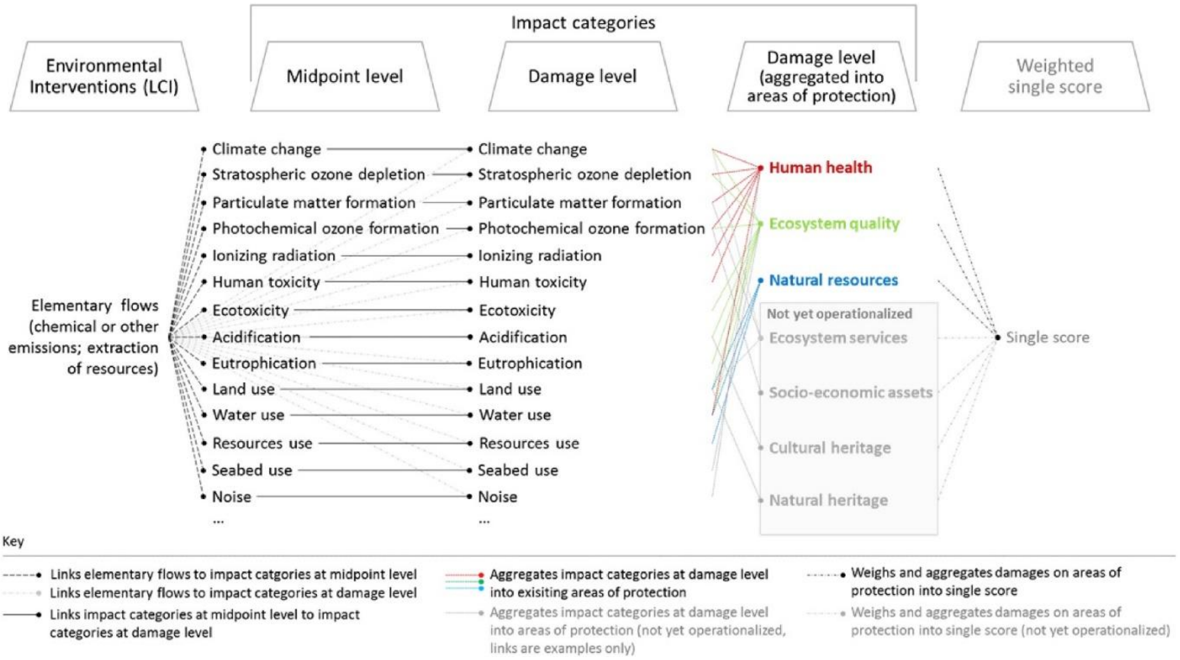


Figure 2: Structure of the LCIA framework. List of impact categories is not exhaustive. Source: Verones et al. (2017)

This part of an LCA denotes the life cycle impact assessment (LCIA), which requires characterization factors (CF) for quantifying impacts to specific categories such as e.g. climate change and ecotoxicity (Figure 2). A CF is the product of a fate factor and an effect factor for a specific stressor induced by an intervention (Hauschild and Huijbregts, 2015) and quantifies the given impact. The fate factor denotes the intensity of a stressor and how is distributed in the environment, while the effect factor represents the increase in a given response per unit increase in pressure (van de Meent and Huijbregts, 2005). Impact categories at the midpoint level may be normalized and further aggregated into areas of protection, currently consisting of the categories human health, ecosystem quality and natural resources (Verones et al., 2017).

In order to quantify the effect of stressors on ecosystems, species sensitivity distributions (SSDs) that quantify the sensitivity of species to a stressor are commonly applied as input data (Hauschild and Huijbregts, 2015). SSD's are created from dose-response modelling, which includes estimating the relationship between the exposure to a stressor and an observed effect. Dose-response curves can take different shapes depending on the given stressor, but when the background concentration of a stressor in the environment is unknown, a linear approach may be adopted (Hauschild and Huijbregts, 2015). The term effect concentration (EC) is used about a specific concentration of a stressor at which a share of the population is affected. A commonly applied EC when measuring the toxicity of a stressor is the EC50, which refers to the estimated concentration where 50% of a given population is observed affected (Traas and van Leeuwen, 2007).

2.3.1 LCIA and mismanaged plastic waste

The marine environment is an underrepresented ecosystem in the current standard LCA framework (Woods et al., 2016), with only two directly related impacts categories, namely marine eutrophication and marine ecotoxicity (ReCiPe, 2009). In the absence of an impact category accounting for mismanaged plastic waste, the trade-offs associated with the extensive use and careless disposal of plastics today cannot be fully quantified, resulting in an underestimation of the environmental impact (Schwarz et al., 2019). Contrarily, the use of plastic as a material may often be associated with lower environmental impacts when considering other impact categories only, such as climate change, as direct CO_2 emissions from transport can be substantially reduced when using a lightweight material as plastic (Humbert et al., 2009). However, the littering aspect of mismanaged plastics are unaccounted for in these assessments, which implies assuming a 100% waste collection rate, where all the plastic goes to recycling, incineration or landfill (Boucher et al., 2019). This is an unrealistic assumption as losses and mismanagement of plastic is known to happen throughout the life cycle of plastics (Ryberg et al., 2019), and adverse impacts on marine fauna caused by this material are prevalent (Wilcox et al., 2016). As such, the development of impact categories accounting for the effects of mismanaged plastic debris on marine ecosystems is warranted, and entanglement in macroplastic debris represents one potential pathway to marine biodiversity loss.

3 Methodology

3.1 Standardizing entanglement rates

An existing database of entanglement rates collated by McHardy (2019) provided the baseline and determined the subset of species for this work. As the database included entanglement rates derived using various approaches, all primary sources were re-assessed in an attempt to harmonize the rates. The percentage of a population entangled annually was deemed the preferred entanglement rate type for standardization. This entailed a conversion of several rates, such as those based on the number of stranded individuals rather than a population size. Updating the rates was achieved by researching which population(s) that a given entanglement rate is likely to pertain (section 3.2), followed by obtaining site-specific population estimates from scientific literature and wildlife monitoring reports. This was required for three entanglement rates in the current dataset; the Common Bottlenose Dolphin (*Tursiops truncatus)*, the Florida Manatee (*Trichechus manatus latirostris)* and the North Atlantic Right whale (*Eubalaena glacialis*). For the latter species, the count of affected individuals was also replaced as updated information was available (Pettis et al., 2020).

Entanglement rates derived from the share of observed entangled individuals in relation to a systematic count of all individuals over a larger area were deemed robust, and did not require a conversion by a calculated population estimate from literature. This was the case for two species in the current work, namely the Northern Gannet (*Morus bassanus*) and the Grey Seal (*Halichoerus grypus*). Finally, all rates were divided over the number of survey years, resulting in population specific annual entanglement rates.

In the re-assessment of the primary sources of entanglement rates, datapoints that were deemed too uncertain in representing a population size were omitted from this report. This included entanglements rates obtained from a small sample size or inconsistent observer effort compared to other datapoints pertaining the same species. Moreover, datapoints were excluded from further analyses if the rates obtained from a secondary report did not match the underlying data in the primary source, and if the data could not effectively be harmonized with the rest of the subset.

3.2 Delineating potential exposure areas

3.2.1 Approach

A general approach for delineating the potential exposure areas is summarized in the following. Details on the rationale and the value choices made for each population-specific potential exposure area follows in section 4.2 of Results and Discussion.

As a starting point for each species, IUCN (2020) distribution maps and OBIS (2020) Seamap observation points were used to obtain an understanding of general distribution patterns. Following this, extensive literature analyses using the databases ISI Web of Knowledge and Google Scholar were conducted for each (sub)species, with a focus on the ecology of the species within the regions of where the entanglement data was obtained. In addition, migratory patterns were considered, in order to assess the year-round distribution and exposure to plastic debris.

In assessing which areas are most heavily utilized by a population, foraging ranges were selected as the primary focus, as this reflects daily activities of which marine animals may come in contact with plastic debris at sea (Thiel et al., 2018). Other commonly used terms to describe the distribution of populations such as home and core range (Fifield et al., 2014) were also included in the assessment. An attempt to obtain representative mean or core ranges was made, as these reflect high use areas for the majority of the population, and are as such linked to a greater temporal exposure to plastic debris. As such, population-specific mean foraging ranges measured by tracking devices on several individuals over multiple years were deemed the ideal basis for delineating the potential exposure area.

Some entanglement rate estimates are given on a population level, while other sources specify which age/sex classes that were most frequently observed entangled. Likewise, some rates pertain a specific season, while others are obtained from year-round surveys. As such, ranges specific for demographic groups within a population or season were averaged or used separately depending on requirements to match the entanglement data, and what was available from the literature. Unless specified to season and/or age and sex classes in the specific section (4.2), the entanglement rates concern the annual exposure of the whole population. In addition, for species where a maximum foraging or home range could be distinguished from the mean, an extended range was delineated for the purpose of sensitivity testing (section 3.3.3).

Data at the level of specificity and robustness described above does not exist for all populations of species. Owing to this, range data obtained from observational local surveys were chosen over more general species tracking data when the literature suggested large discrepancies in the ranges among different geographic populations of a species. In addition, delineation of ranges based on bathymetry was done using data from GEBCO (2020) for species when appropriate. This was mainly done for species that are not colonial or central-place foragers and thus not tied to specific sites which allows mean foraging ranges to be estimated. Bathymetry data was used to delineate areas in combination with estimated abundance maps obtained from literature.

Moreover, for wide-ranging migratory species (e.g. the cetaceans) where mean ranges and distinct geographic boundaries between populations could not be ascertained, a consideration of the circumstances of the entanglement incidents was made. When the level of detail in the entanglement data and that obtained from the literature analysis allowed for it, this served as an indication of the possible distances between the area of exposure and the area where the entanglement was observed. In most cases, it is assumed that entanglements to some degree will hamper the mobility and migration potential of individuals, and the range can as such be narrowed down to a specified region. Similarly, nearby sources to plastic debris such as regional fisheries operations were considered for the same reasons if the entangling material was detailed in the primary sources and could be attributed to specific sources.

Potential exposure areas were delineated for all species populations with associated standardized entanglement rates. In addition, for one species where the entanglement rate could not effectively be standardized, the Common Minke Whale (*Balaenoptera acutorostrata)*, it was nevertheless deemed relevant to delineate a potential exposure area for future modelling purposes, as the literature indicate that the specific population is prone to entanglement-related mortality (Northridge et al., 2010).

In the following sections, the general term "range" is also used to denote the potential exposure areas of species populations, unless otherwise specified. All ranges were modelled using ArcGIS® software by Esri.

3.2.2 Assessment of robustness

An assessment of the mean potential exposure areas was made by assigning levels of confidence (low, moderate, high) based on the underlying data applied to delineate the areas. The criteria for the confidence levels involved the robustness of the methods applied in studies for estimating ranges, and to what degree the available data was representative of the specific population of which the entanglement data concerned. As a combination of several sources influenced the delineation of areas, only the most relevant data source of highest robustness obtained for the specific population was assessed. The criteria applied for assigning confidence levels are detailed in Appendix 1. As the mean potential exposure areas were to be considered the primary focus of this thesis, the extended areas defined for the purpose of sensitivity testing were not included in the robustness assessment.

3.3 Coupling of entanglement rates to plastic debris exposure

3.3.1 Modelling plastic debris exposure

Estimates of the spatial distribution of marine plastic debris was provided by Eriksen et al. (2014) as the mass of floating plastic debris per square kilometer (g/km^2) with a 0.2 decimal degrees grid-cell resolution. The data used for the model estimations are based on expeditions over the period 2007-2013 (Eriksen et al., 2014). In their model, plastic debris abundance was originally classified by four size categories; two microplastic, one mesoplastic (4.76-200 mm) and one macroplastic (>200 mm). As microplastic is generally not considered a relevant size dimension for entanglement, the meso- and macroplastic categories were merged to encompass all plastic debris >4.75 mm (Woods et al., 2019), following a recommended definition of macroplastic as >5 mm (GESAMP, 2019).

Plastic debris concentrations (g/km^2) were calculated for each potential exposure area by using the "Spatial analyst: zonal statistics" tool in ArcGIS. This was done in order to link the standardized entanglement rates, i.e. the estimated prevalence of entanglement, to estimated plastic concentrations within the delineated geographical areas. Plastic debris concentrations (g/km^2) were calculated using both the mean of all cells and the maximum cell value found within the delineated areas. Summarizing the plastic concentrations as the mean was deemed most relevant for the current concentration-response modelling approach, while the maximum values were derived for the purpose of sensitivity testing (section 3.3.3).

3.3.2 Dose-response calculation

In order to gain insight into how sensitive the populations in the current dataset are to macroplastic debris entanglement, dose-response modelling was conducted. As the relation between concentrations of macroplastic debris and observed entanglement effects is not empirically known, the dose-response model calculations in this thesis followed a linear approach. This involved assuming a zero percent entanglement rate when the concentration of plastic debris is zero (Woods et al., 2019). Moreover, as all but one of the standardized entanglement rates in the current dataset were below 5%, an EC5 value was considered the most appropriate fit for the dose-response modelling. Applying EC5 instead of the commonly used EC50 minimizes the extrapolation from the entanglement rates in the current dataset to the EC.

In most cases, only one population-specific entanglement rate and associated potential exposure area was available as datapoints for extrapolating species-specific EC5's. For species where distinct potential exposure areas associated with different entanglement rates were identified, the datapoints could be combined in the linear regression. Exceptionally, two separate EC5 values were derived for the Northern Gannet (*M. bassanus*) owing to distinct differences in breeding and non-breeding behavior and distribution (detailed in 4.2.1). To model the gannet's overall year-round plastic debris sensitivity, the two EC5's were finally combined in a weighted average; the breeding area specific sensitivity being weighted by 60% and the wintering by 40% owing to difference in the average time spent at each location in their annual cycle (Garthe et al., 2017b, Kubetzki et al., 2009, Fort et al., 2012).

Preliminary EC5 values were derived for all species, making possible a ranking of the species according to their sensitivity, measured by the plastic debris density (g/km^2) at which 5% of the population is annually entangled. Additionally, EC5 values linked to the extended ranges of species and maximum plastic debris concentrations were also derived for sensitivity testing purposes.

3.3.3 Testing sensitivity to modelling choices

The estimated exposure concentration within an area can be expected to differ depending on the geographical extents of the delineated area. As such, an attempt to gain a first indication as to how sensitive the model for developing the effect factor might be to range choices was made. This was tested by comparing how the ranking of species according to the EC5 changed when applying an additionally defined extended range.

Extended ranges were defined for species where linked uncertainty to the derived core or mean range was present, and a distinction could be made between this and a maximum range (detailed for the relevant species in section 4.2). For the remainder of the populations, the IUCN (sub)species range was applied as the extended range for testing purposes. For species where spatial population segregation could be established, the IUCN ranges were further modified to only cover the general region of the entanglement data (See Appendix 2 for brief descriptions). In addition, choice of summary statistics for calculating the plastic concentrations within the potential exposure areas was tested for its influence on the model. Combining the choice of mean or extended exposure area with mean or maximum plastic debris concentrations (g/km^2) yielded a total of four combinations for each population associated with an entanglement rate. This made possible a comparison of how the sensitivity ranking changed depending on which modelling combination that was applied.

4 Results and Discussion

4.1 Standardized entanglement rates

A re-assessment of the entanglement database lead to a selection of 10 marine species with an associated 15 standardized entanglement rates. The rates have been updated to represent the estimated share of a population that are annually entangled in macroplastic debris, but vary in terms of how the observations of entanglement were made, the endpoint effect and the duration and timing of the study period (Table 1).

Table 1: Standardized entanglement rates and associated method, endpoint effect and time period of observations. Entanglement rate method: E = Number of observed entangled individuals, E_S = Number of stranded entangled individuals, P_T = Total population estimate, P_C = Population as number of counted individuals.

Species	Rate (%)	Entanglement rate method			Endpoint		Time period		Source
		E/ P_C	E / P_T	E_S / P_T	Chronic entanglement	Mortality	Season specific	> 1 year	
Northern Gannet *(M. bassanus)* Breeding	3.05	-	X	-	X	-	X	X	Werner et al. (2016)
Northern Gannet *(M. bassanus)* Non-breeding									
Gulf of Cadiz	0.11	X	-	-	X	-	X	-	Rodriguez et al. (2013)
Cantabrian Sea	0.36	X	-	-	X	-	X	-	Rodriguez et al. (2013)
Mauritania	20.16	X	-	-	X	-	X	-	Rodriguez et al. (2013)
Common Guillemot *(U. aalge)*	1.05		X	-	X*	-	X	-	Werner et al. (2016)
Northern Fulmar *(F. glacialis)*	1.80		X	-	X*	-	X	-	Werner et al. (2016)
Antarctic Fur Seal *(A. gazella)*									
Signy Island	0.02	-	X	-	X	-	X	X	Waluda and Staniland (2013)
Bird Island	0.016	-	X	-	X	-	-	X	Waluda and Staniland (2013)
Cumberland bay	0.09	-	X	-	X	-	-	X	Waluda and Staniland (2013)
Australian Fur Seal *(A. p. doriferus)*	0.014	-	X	-	X	-	-	X	Lawson et al. (2015)
Common Bottlenose Dolphin *(T. truncatus)*	0.08	-	-	X	-	X	-	X	Adimey et al. (2014)
Florida Manatee *(T. m. latirostris)*	0.46	-	-	X	-	X	-	X	Adimey et al. (2014)
Grey Seal *(H. grypus)*	4.30	X	-	-	X	-	-	X	Allen et al. (2012)
North Atlantic Right Whale *(E. glacialis)*	1.88	-	X	X	X	X	-	X	Pettis et al. (2020)
Steller Sea Lion *(E. j. monteriensis)*	0.26	-	X	-	X	-	X	X	Raum-Suryana et al. (2009)

* Primary reference is in press and endpoint effect could not be confirmed.

4.2 Potential exposure areas

Potential exposure areas were delineated for a total of three seabird species and eight mammals (Figure 3).

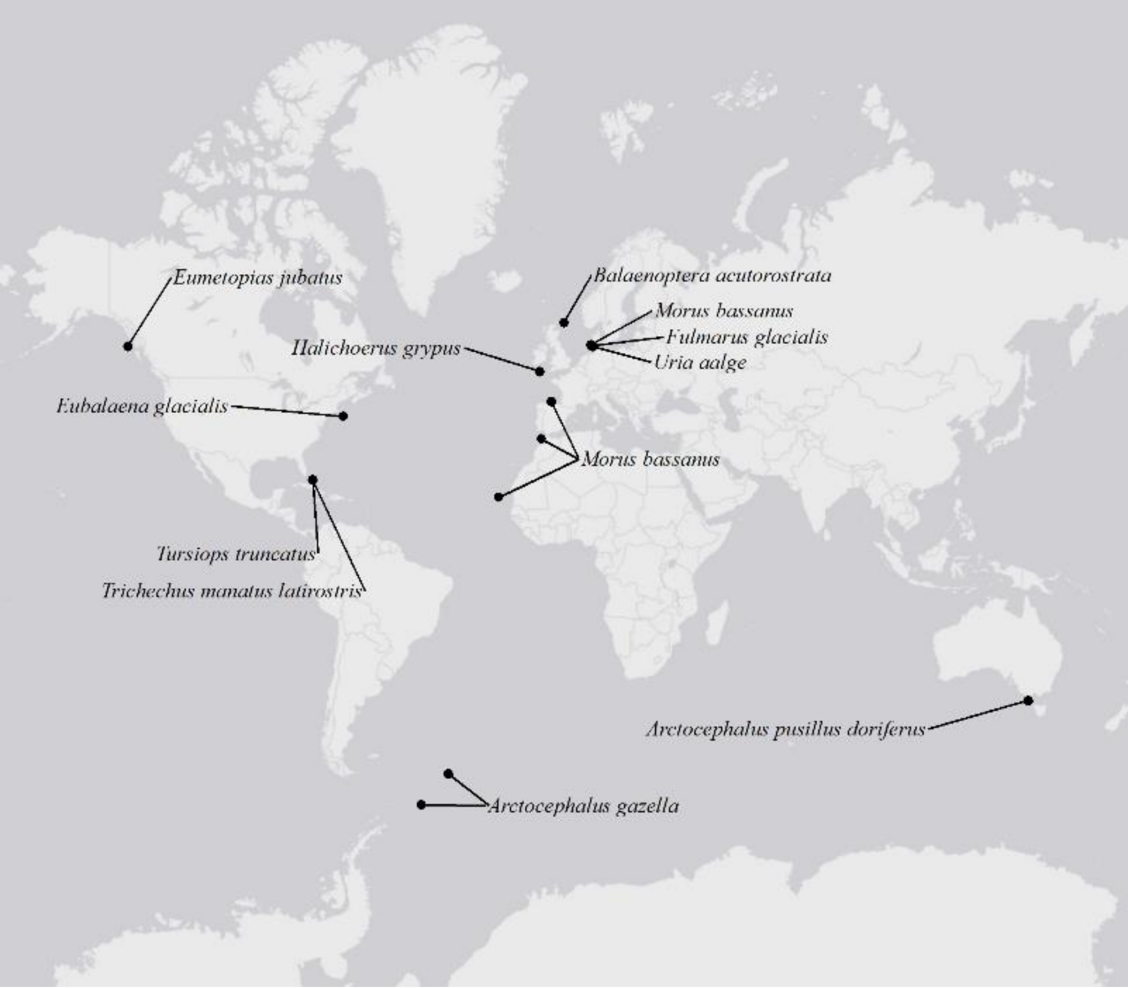


Figure 3: The general location of datapoints for collected species entanglement rates of which potential exposure areas were delineated.

4.2.1 Northern Gannet (*Morus bassanus*)

The Northern Gannet (*M. bassanus*), hereafter gannet, can get entangled in plastic debris when they plunge-dive for fish, either accidentally or by confusing it for prey (Schrey and Vauk, 1987, Conradt, 2016). In addition, breeding gannets and their chicks are subject to a specific type of plastic exposure in the nests distinct from that at sea during the rest of the year. Breeding gannets also act as central place foragers, which restricts them to a considerably smaller foraging range during this period compared to the rest of the year (Hamer et al., 2000). Therefore, it is deemed useful to keep the breeding- and wintering specific entanglement rates and potential exposure areas separate for this species at this stage of the modelling (McHardy, 2019)

4.2.1.1 Breeding at Heligoland

Entanglement data

Entanglements were registered during the breeding season of 2014 and 2015, on the German island Heligoland, which holds the only gannet colony in the south-eastern part of the North Sea (Garthe et al., 2017a). Breeding gannets select plastic debris floating at the sea surface and incorporate it in their terrestrial based nests (Tasker et al., 2000). This causes entanglements of primarily their feet and legs, but also wings (Votier et al., 2011). Averaged over the two sampling years, the annual nest entanglement rate was 3.05% (Werner et al., 2016). Both adults and nestlings are susceptible to this type of entanglement (Montevecchi, 1991), although nestlings are more prone and likely to die from the injuries (Votier et al., 2011).

Potential exposure area

During the breeding season, the gannets nest in large colonies on cliffs and offshore islands (BirdLife International, 2018b). Although the entanglements are occurring in the nests and not directly at sea, the marine space they use to collect nest material may be applied as the potential exposure area. They collect material primarily from the marine environment, and the plastic in their nests has been proposed as an indicator of the abundance of plastic debris in their local marine environment (Montevecchi, 1991, Bond et al., 2012). Although nestlings are more prone to nest entanglements than adults (Votier et al., 2011) they do not leave the nests during this period. Owing to this, ranges based on movements by adults are suitable for determining the exposure area.

To the maximum extent, birds may travel 500-700 km away from the colonies to forage (Hamer et al., 2000, Thaxter et al., 2012, Wakefield et al., 2013), but they mainly stay within continental shelf areas (Mendel et al., 2014). Tracking of gannets from the colony on Heligoland during the breeding season of 2015 showed that they fly in all directions to forage, mainly at distances of 3-80 km away from the island (Garthe et al., 2017b), and more often to the west and northwest of the island (Garthe et al., 2016). In addition, they generally make shorter trips in the vicinity of the colony to gather nest material (Garthe et al., 2017b). This is supported by Pettex et al. (2012) who observed gannets from Norwegian colonies only returning with nest material when trip durations were less than 30 minutes. As such, the larger extent of foraging trips may be excluded from the range of nest material collection, and a narrow range specific to collection of nest material in close proximity to the colony can be applied. As there is a lack of knowledge on the extent that seabirds travel to gather nest material (Thompson et al., 2020), assumptions are required. Pettex et al. (2012) found that northern gannets had an average flight speed of 50 km per hour over three years of tracking. As trips of < 30 minutes duration were considered nest material collection trips, it can be derived that they could extend to a distance of 12.5 km from the colony when accounting for the return trip. As such, a range of 12.5 km extending in every direction from Heligoland is applied as the potential exposure area (Figure 4).

Extended range

As gannets are opportunistic in nature, it cannot be ruled out that they may pick up nesting material on foraging trips. O'Hanlon et al. (2019) assessed 29 gannet colonies, including Heligoland, and found that the presence of plastic debris in nests was positively related to recent years' fishing effort within 100 km of the colony. Taking this into account, a range of 100 km can be delineated (Figure 4), which agrees with the maximum distance travelled for the majority of foraging trips from Heligoland (Garthe et al., 2017b). Moreover, it is also in line with a global mean foraging range (92.5 km) delineated for this species from multiple direct studies of gannet colonies (Thaxter et al., 2012).

Figure 4: Mean and extended potential exposure area for the Northern Gannet at the breeding colony of Heligoland, as indicated by legend.

4.2.1.2 Wintering areas

Entanglement data

Entanglement rates for non-breeding gannets in wintering areas were given as 0.11%, 0.36% and 20.16% for the Gulf of Cadiz, Cantabrian Sea and the coast of Mauritania, respectively (Rodríguez et al., 2013).The surveys from the Spanish coast were conducted during the wintering season, while the observations from Mauritania were made in June. As most adult birds will be at their breeding colony during the summer, the presence of adult birds at the Mauritania site was very low (Rodríguez et al., 2013). This may aid in explaining why this site has an elevated entanglement rate compared to the two other sites. There was only one adult observed entangled at all sites combined, indicating that juveniles that are more inexperienced in foraging are more susceptible (Rodríguez et al., 2013).

Potential exposure area

Gannets from all east Atlantic colonies are assumed to overlap in the wintering areas (Wanless, 2002). Additionally, individuals from the western Atlantic have been found to winter off West Africa (Garthe et al., 2016), albeit not considered as a widespread strategy as they are birds of continental shelf waters and will not often cross the Atlantic (Fort et al., 2012, Fifield et al., 2014). As such, this part of their range can be excluded from the potential exposure area.

A factor that affects the extent of the potential exposure area is the mobility of entangled gannets, as the population migrates considerable distances from the colonies to the wintering areas where the entanglements were registered. Gannets have been observed capable of flying with plastic entanglements around the bill or other parts of their body (Schrey and Vauk, 1987, Rodríguez et al., 2013), suggesting that the relevance of their whole range as a potential exposure area needs to be considered. As it is not feasible to ascertain which European colonies the entangled birds in the wintering areas originate from (but see Fort et al., 2012) this would encompass a large area including migration routes from all colonies. As gannets are strong flyers, this entails the inclusion of vast areas that they merely fly over during the migration, and does not represent where they are spending most of their time in contact with the sea and plastic debris.

The possibility of the entanglements occurring already on their breeding grounds as nest entanglements can be considered less probable, as birds in the wintering areas were observed with plastic debris around their bills only (Rodríguez et al., 2013) rather than other parts of their body which would be expected from nest entanglements (Votier et al., 2011, Montevecchi, 1991). Moreover, the current entanglement data includes observations of individual birds with bill deformations due to plastic entanglement (Rodríguez et al., 2013), suggesting that the injuries may be severe which can exert a restricting effect on their energy budgets and migration capacity. Furthermore, it has been suggested that gannets with plastic entangled around their bills will not be able to liberate themselves (Schrey and Vauk, 1987), which would results in starvation and render a long-distance migration unlikely. Following this, it can be implied that the gannets get entangled in plastic debris after arrival at the wintering areas.

The general foraging range of non-breeding gannets is assumed to be large, as they are not constrained by the need to return to their nestlings (Gaston, 2004). Given the practical difficulties with observing seabirds away from their breeding colonies (Fifield et al., 2014), there exists little data on the foraging range of gannets outside the breeding season. However, they are mainly registered over the outer continental shelf areas, and rarely in the deep ocean beyond the 200 m depth contour (Camphuysen and van der Meer, 2005, Grémillet et al., 2015). Moreover, a multiple year tracking study on wintering gannets in the Western Atlantic found the radius of the mean home range to be 292 ± 45 km, and the mean core range 135 ± 21 km (Fifield et al., 2014). Owing to a lack of wintering specific ranges for the eastern Atlantic population, this will be applied for all three wintering locations (Figure 5), extending out from an approximation of the survey area specific to each wintering site (Rodríguez et al., 2013).

Extended range

As the European gannet population occupies wintering locations along a gradient (Garthe et al., 2012) discrete boundaries between the different wintering sites do not exist. As such, an extended range using the mean home range of 292 km including a connection between all three wintering locations can be defined (Figure 5). By applying this connected range, the site-specific entanglement rates are also averaged to one combined rate for non-breeding gannets.

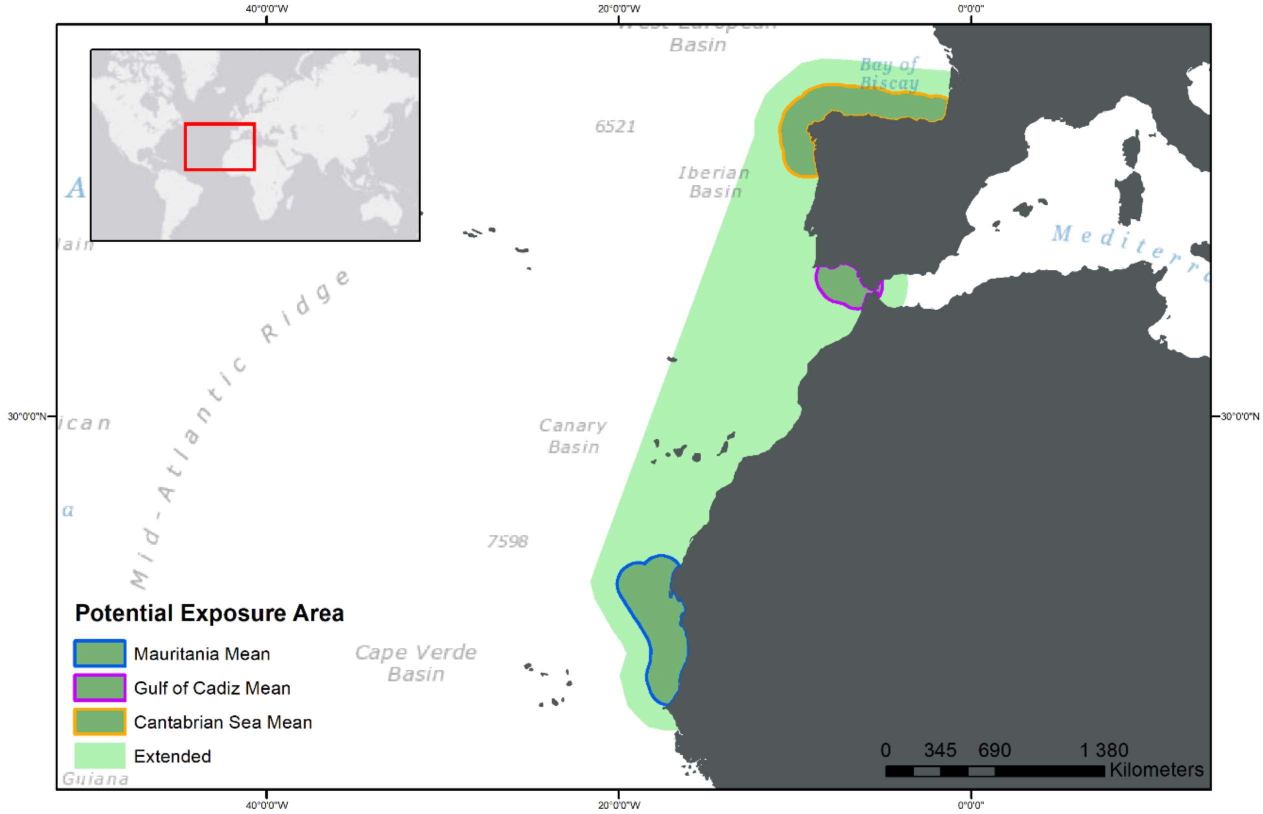


Figure 5: Mean and extended potential exposure areas for the Northern Gannet at the wintering sites off Spain and Mauritania. Mean potential exposure areas for the specific wintering locations as indicated by legend.

4.2.2 Common Guillemot (Uria aalge)

Entanglement data

Entanglements of the Common Guillemot (*U. aalge)*, hereafter guillemot, were registered on Heligoland in 2014 and 2015 (Werner et al., 2016). The season of sampling was not specified for the data pertaining the guillemots, but as the rate was obtained from the same source as that of the Northern Gannet (*M. bassanus)* during the breeding season on Heligoland, it is assumed that the sampling was done during the same time period. Averaged over the two sampling years, the annual entanglement rate was 1.05% (Werner et al., 2016).

Potential exposure area

Guillemots are pelagic seabirds that only go on land during the breeding season, when they nest on steep cliffs (Mendel et al., 2014). Guillemots do not build nests themselves, but they reside on Heligoland alongside Northern Gannets (*M. bassanus*) and are known to get entangled in the plastic from these nests as they take off from the cliffs (Conradt, 2016). This is increasing their exposure to plastic at the breeding sites, but the contribution of this relative to entanglements occurring during activity at sea has not been quantified. As such, no distinction between entanglements in nest and at sea is made for this species.

Heligoland is the only German breeding site for this species, located at a considerable distance from other North Sea Guillemot colonies (Sonntag and Hüppop, 2005). As such, it can be assumed that mainly natal birds are observed there during the breeding season. Guillemots from other colonies are found in German waters as well, but this is more common outside the breeding season (Mendel et al., 2014, Markones et al., 2010). Moreover, the wintering range of guillemots will not be considered as the available data is assumed to have been collected during the breeding season.

The foraging range of guillemots is assumed to be restricted to the vicinity of the colony on Heligoland because they are not strong flyers (Leopold et al., 1995) and mainly swim when foraging (Markones et al., 2010). Observations of guillemots on Heligoland have shown that they are most abundant in the waters surrounding the colony and that the density drops with increasing distance (Leopold et al., 1992). Flight and hence long foraging trips are costly due to their high wing-loading, which also incentivizes them to travel directly to feeding areas that are more or less predictable (Gaston, 2004).

Their foraging range has been assessed using ship based and aerial surveys, which can be justified as the distance to the nearest colonies is greater than the range of guillemots during the breeding season (Dierschke et al., 2004). From these observations, a mean foraging range radius around Heligoland of up to 10 km was found (Leopold et al., 1995, Dierschke et al., 2004), which is in agreement with a two year telemetry tracking study from a colony in Shetland (Monaghan et al., 1994). Dierschke et al. (2004) also found that most birds were sighted within 15-20 km, further affirming their overall narrow range. Although the home range of guillemots are subject to interannual variation, their core habitat locations are less influenced by environmental factors and can thus be expected to be more consistent (Tranquilla et al., 2014). As the Heligoland specific estimates are based on visual surveys and not recent tracking data, the higher value of 20 km is selected to represent their potential exposure area during the breeding season (Figure 6).

Extended range

An average range of 37.8 km for guillemots has been derived based on several studies from west and east Atlantic colonies (Thaxter et al., 2012), although it can be expected that the guillemot population on Heligoland maintain a smaller range than this because of the small size of the colony. Nevertheless, this average will be applied as the extended range for the guillemots at Heligoland (Figure 6).

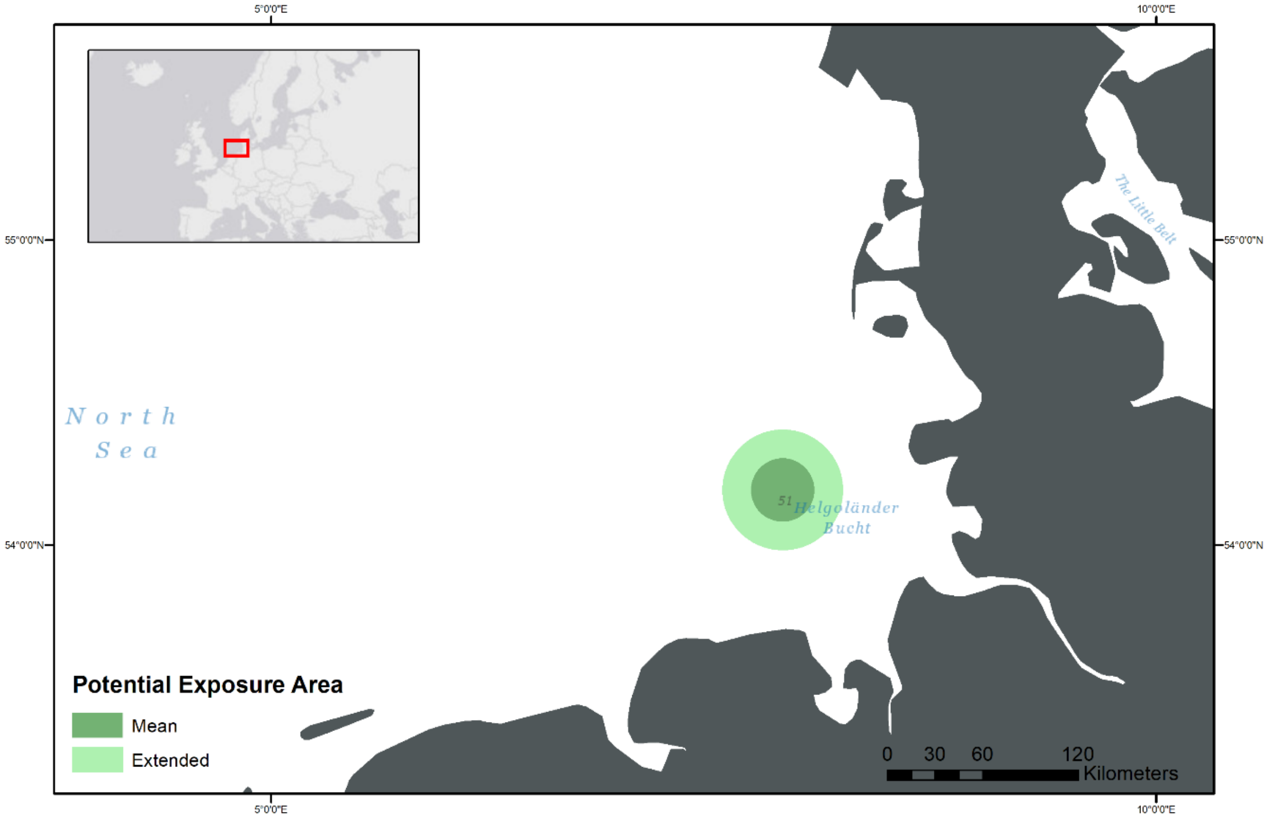


Figure 6: Mean and extended potential exposure area for the Common Guillemot at the breeding colony of Heligoland, as indicated by legend.

4.2.3 Northern Fulmar (*Fulmarus glacialis*)

Entanglement data

An annual entanglement rate for the Northern Fulmar (*F. glacialis),* hereafter fulmar, residing on Heligoland was found to be 1.80% (Werner et al., 2016). The season and year of sampling was not specified for the data pertaining the fulmars, but as the rate was obtained from the same source as that of the Northern Gannet (*M. bassanus)* during the breeding season on Heligoland, it is assumed that the sampling was done during the same time period.

Potential exposure area

Fulmars are wide ranging and highly abundant throughout large parts of the North Atlantic and North Pacific, including regions in the high Arctic (BirdLife International, 2018a). The European population is however considered endangered (BirdLife International, 2015).

As strong flyers, they can increase their flight speed in order to cover larger areas in search of food (Elliott and Gaston, 2005) and may forage across several north-east Atlantic management areas during the breeding season (Edwards et al., 2016a). Movements of individuals across the Atlantic occurs (Mallory et al., 2012), and trips to the Mid-Atlantic Ridge during the breeding season have been registered, but this is unlikely to be a widespread strategy (Edwards et al., 2013). Such flights are energetically costly and seabirds will thus often choose to stay within certain foraging areas (van Franeker and Law, 2015). The local waters around colonies (<400 km) are generally considered as most important for breeding success (Cordes et al., 2015).

The German bight, which is used by the Heligoland fulmars, is also used by fulmars from other European colonies (Garthe, 2019). However, it is assumed that the entanglement data is collected during the breeding season, and this species tends to breed at their natal or neighboring colonies (Burg et al., 2003). With a distance of several hundred kilometers to other colonies, it can be assumed that birds observed on the Heligoland island during the breeding season are local. The species generally favors continental shelf areas (Weimerskirch et al., 2001), and Northern fulmars in German waters appear to avoid the coastal areas and rather concentrate in the central North Sea water (Camphuysen and Garthe, 1997, Garthe et al., 2009).

The variation in foraging ranges is large for this species (Mallory et al., 2012), and considerable discrepancies are found even between colonies of similar size in the north eastern Atlantic (Hamer et al., 1997), as well as within colonies during different phases of the breeding season (Weimerskirch et al., 2001, Ojowski et al., 2001). As such, there is a need for considering the local conditions of the colony of interest, but estimates of foraging ranges specific to northern fulmars of Heligoland could not be obtained. Thaxter et al. (2012) suggested, based on a literature review, a mean foraging range of 47.5 km, and a mean maximum range of 400 km for the species. This range is derived from studies of colonies at a higher latitude of the fulmars' range (Greenland, Svalbard and Shetland) and cannot be extrapolated to Heligoland without caution. The range is however useful as a mean value for the species in general, as all values from each colony have been averaged in order to minimize the effects of density-dependence and intra-specific competition that may lead to differences in the ranges of small vs. large colonies (Thaxter et al., 2012).

Heligoland is home to one of the smallest populations of fulmars with an estimated 107 breeding pairs (BirdLife International, 2015). As smaller colonies tend to be associated with smaller foraging ranges (Gaston, 2004), the use of the mean value of 47.5 km is considered sufficient in encompassing the range where the fulmars will be most abundant during the breeding season (Figure 7).

Extended range

As visual surveys of the German bight illustrate that fulmars in general are more abundant in the central North Sea rather than continental coast (Garthe, 2019), it cannot be ruled out that fulmars from Heligoland may also use these areas. As such, the mean maximum range of 400 km (Thaxter et al., 2012) can be applied as the extended range to also encompass this area (Figure 7).

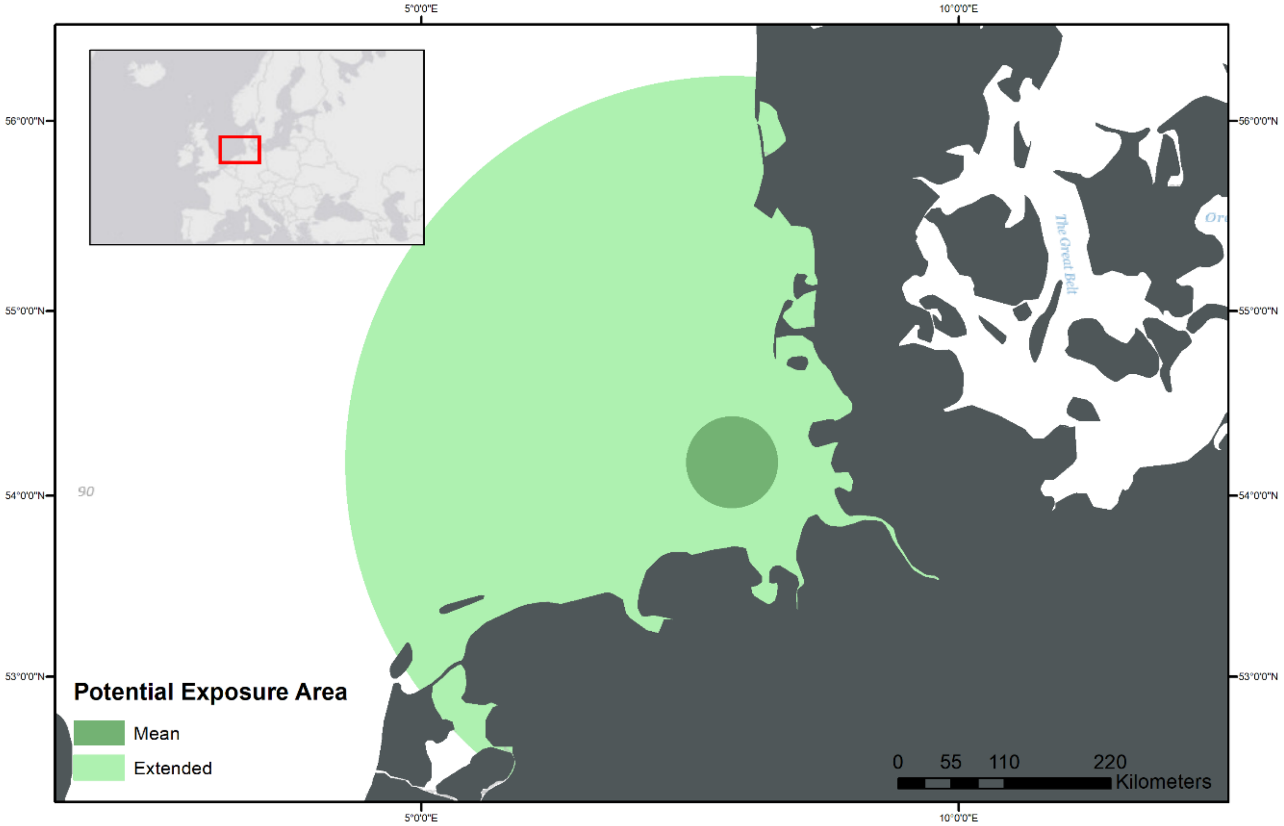


Figure 7: Mean and extended potential exposure area for the Northern Fulmar at the breeding colony of Heligoland, as indicated by legend.

4.2.4 Antarctic Fur Seal (*Arctocephalus gazella*)

Entanglement data

Entanglement data for the Antarctic Fur Seal (*A. gazella),* hereafter fur seal, was obtained from three locations on islands in the Scotia Arc during the austral winter and summer from 1995-2013 (Waluda and Staniland, 2013). Entanglement rates of 0.02%, 0.016% and 0.09% were registered for the populations at Signy Island (South Orkney), Bird Island and Cumberland bay (South-Georgia) respectively. Juvenile male seals were found to be most frequently entangled, but all demographic groups were affected (Waluda and Staniland, 2013). In the majority of the observations, fishing related debris such as packaging bands, line and fishing net were tightly encircling the neck of the seals (Waluda and Staniland, 2013).

Potential exposure area

Fur seals that inhabit the Southern Ocean are suggested to have a wide range without distinct subpopulations (Hofmeyr, 2016). Although this species is distributed and found in smaller numbers on colonies throughout the Southern Ocean, only the entanglement data specific colonies will be considered for delineation of the potential exposure area, as both genders show high levels of breeding site fidelity (Bonin, 2012).

Females act as central place foragers during the breeding season, staying within 350 km of the breeding site during pup rearing (Boyd et al., 1998). During the austral winter, females are free to disperse widely and a tracking study found the mean maximum distance travelled from the colony to be 1259 km per trip (Arthur et al., 2015). However, other studies have found that females from South Georgia remained within 510 km of the colony for most parts of the winter (Staniland et al., 2012). This is in agreement with tracking of females from Bird Island, where factors such as <500 km distance from the colony as well as water depths of <2000 m best predicted where the seals spent the most time, also during winter (Arthur et al., 2017). As the distance between the two study locations on South Georgia is well within this range (approximately 100 km), it is deemed reasonable that seals may travel from one site to another and that they are subject to the same plastic debris exposure. As such, the entanglement rate from these two locations is averaged to 0.053%.

There is a lack of range tracking data from male fur seals and juveniles in this region. However, it is suggested that males range widely for most parts of the year as they do not take part in parental care (Staniland et al., 2007). They maintain territories during the breeding season whereupon they may migrate from South Georgia to the South Orkney Islands (Boyd et al., 1998). This movement pattern supports that the entanglement data obtained from the two island groups relates to one connected population (Boyd et al., 1998). However, population monitoring from Signy Island has shown that mainly young male seals migrating from South Georgia are found on this island in the South Orkneys, which only supports a low number of females and pups (Waluda et al., 2010). In addition, males exclusively were observed entangled at Signy Island, which was only surveyed during the austral summer, coinciding with the time that males from South Georgia migrate to South Orkney (Boyd et al., 1998). As the entanglement rate from this location relates to a subpopulation, it is deemed reasonable to keep the two rates separate to conserve the potential differences in plastic debris exposure when coupled to the entanglement rate.

Owing to the remote location and small human presence of these Southern Ocean islands, it may be questioned whether it is likely that local waters are significant to their plastic debris exposure. However, decades of surveying at Bird Island (South Georgia) and Signy Island (South Orkney) highlight the prevalence of marine plastic debris on these specific sites (Waluda et al., 2020). As such, the seals are exposed to plastic debris in their local environment, likely carried by currents from distant sources or local shipping (Waluda et al., 2020). Owing to this, a radius of 500 km around the sites at South Georgia and Signy island separately can be applied to encompass their core area (Figure 8). Although this estimate is based on tracking studies of females only (Staniland et al., 2012, Arthur et al., 2017), the observations were derived from the winter season when adult females are also unconstrained by parental care.

Extended range

It is deemed likely that the fur seals may get entangled in plastic debris when on foraging trips far away from their breeding beaches (Hofmeyr et al., 2006, Waluda and Staniland, 2013). Antarctic fur seals take foraging trips to the Patagonian coast of South America (Staniland et al., 2012) and east towards the South Sandwich islands (Arthur et al., 2017) when not under parental constraints. As such, these two areas of more distant foraging sites can be encompassed in an extended range (Figure 8). The range was cut by the boundaries of the species' IUCN distribution map in the southern extent of the area.

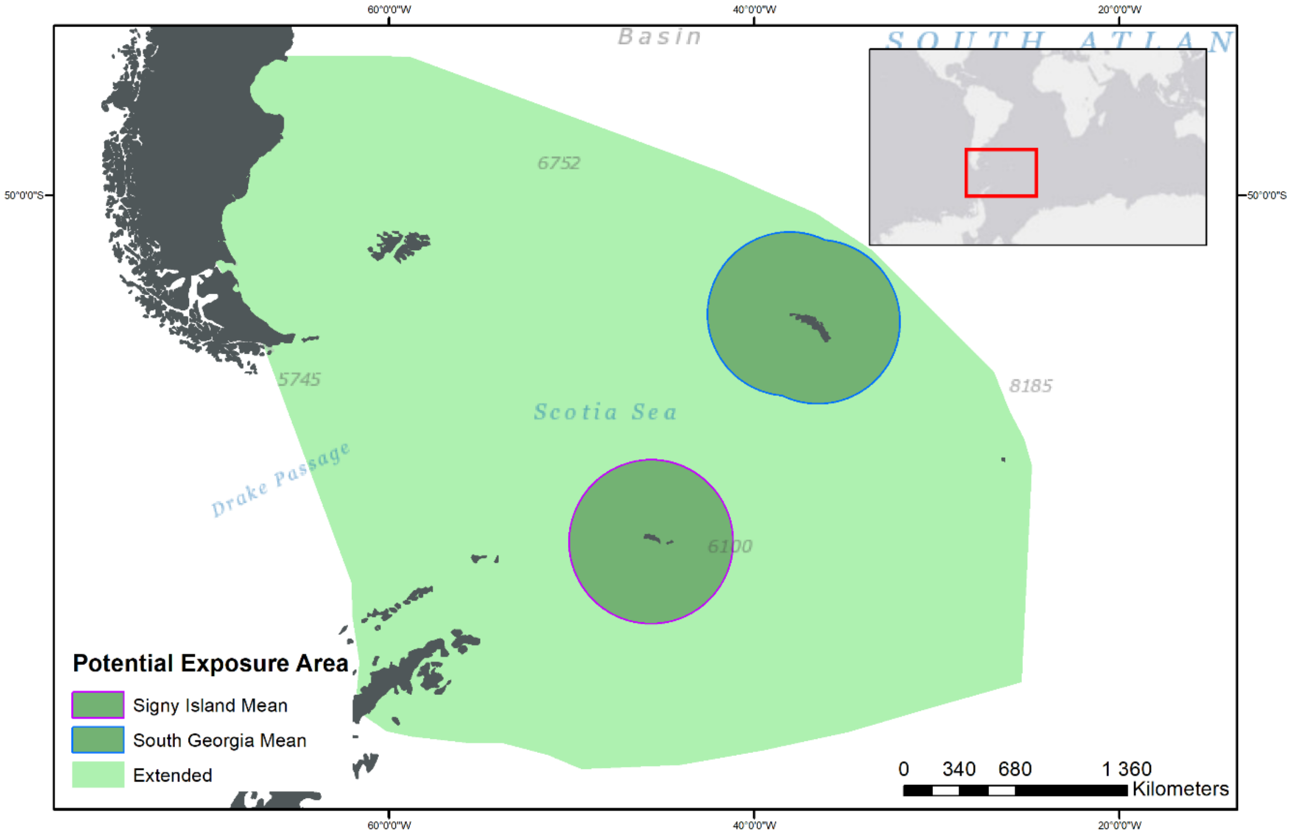


Figure 8: Mean potential exposure areas for the Antarctic Fur Seal around Signy Island (South Orkney) and South Georgia (Bird Island and Cumberland Bay combined), and extended potential exposure area for sites combined, as indicated by legend.

4.2.5 Australian fur seal (*Arctocephalus pusillus doriferus*)

Entanglement data

Entanglements of the Australian Fur Seal (*A. p. doriferus*) found on Seal Rocks in Southern Victoria, Australia was registered every other month between 1997-2012 (Lawson et al., 2015). When accounting for the years surveyed, the annual entanglement rate was found to be 0.014%. Seals in younger age classes were more often observed entangled than adults (Lawson et al., 2015), which is in agreement with other entanglement studies for this species (McIntosh et al., 2015, Pemberton et al., 1992). Fur seals in this region are exposed to several types of marine debris, but entanglements were most commonly associated with materials originating from fishing activities such as plastic twine or rope (Lawson et al., 2015).

Potential exposure area

The Australian Fur Seal (*A. p. doriferus*) is a subspecies of the Afro-Australian Fur Seal, which is found along the south western and southern coast of Africa and in southeast Australian waters (Hofmeyr, 2015). The Australian subspecies inhabits the nearshore waters of the Australian coast from Victoria to Tasmania and across the south of the continent to New South Wales (Lawson et al., 2015). The majority of the population breed and forage in Bass Strait (Kirkwood et al., 2010, Shaughnessy et al., 2010). Seals tagged at the different colonies in the Bass Strait exhibit site fidelity (Kirkwood and Arnould, 2011), but will also frequent other colonies for resting during foraging trips (Arnould and Kirkwood, 2007), indicating that the subspecies makes up one population (Salton et al., 2019, Knox et al., 2017).

The seals forage within the continental shelf (Hoskins et al., 2017), most frequently at depths of 60-80 m (Knox et al., 2017, Arnould and Kirkwood, 2007). Contrary to most other fur seal species they are benthic foragers, but it is unknown to what extent this foraging strategy may affect their likelihood of getting entangled in marine debris (McIntosh et al., 2015). This species has one of the most restricted breeding ranges among fur seals (Arnould and Kirkwood, 2007) and a tracking study found that females nursing pups foraged within 80 km of the colony during winter (Hoskins et al., 2017). Similarly, Kirkwood and Arnould (2011) found that females stayed in areas within 150 km of the colonies, including Seal Rocks, over 90% of the tracking time.

Juveniles are suggested to travel similar distances as adult females (Kirkwood and Goldsworthy, 2013) or shorter (Salton et al., 2019). Males are seen to have a larger core area and foraging range than females and make foraging trips all the way to Southern Tasmania (Salton et al., 2019). This is expected as males are free of parental care constraints, and may range far and use several sites owing to the polygamous breeding behavior of fur seals (Tilzey et al., 2006). However, it has also been found that males maintain a foraging range largely restricted to Bass strait for most parts of the year (Knox et al., 2017). Similarly, other tracking studies have suggested that a majority of males stay within 200 km (Knox et al., 2018) and 300 km (Kirkwood et al., 2007) from their colonies. As such, a radius of 200 km extending from Seal Rocks (Figure 9) may be representative as a mean of the foraging range for females, juveniles and most males.

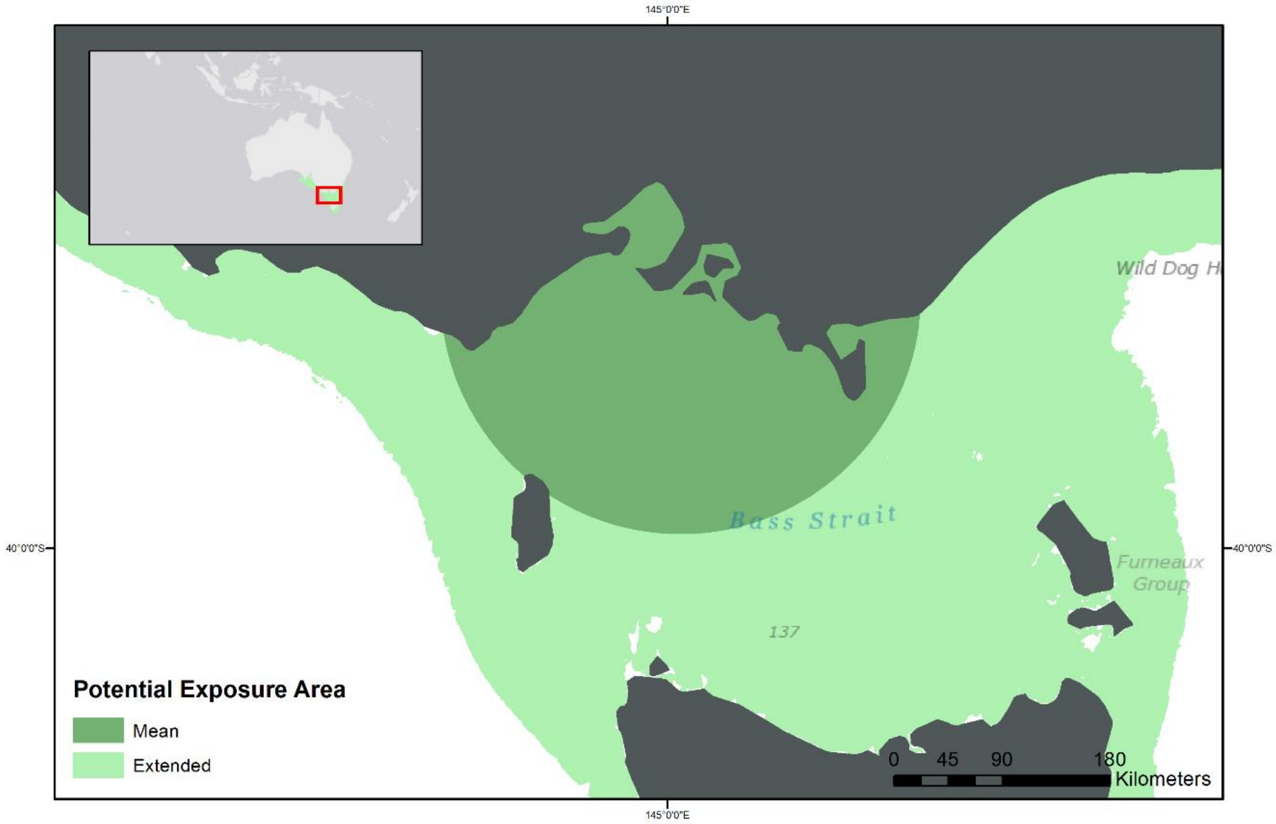


Figure 9: Mean potential exposure area for the Australian Fur Seal, as indicated by legend. Extent of the extended potential exposure area, which is a modified IUCN subspecies range, is shown in the inset map.

4.2.6 Common Bottlenose Dolphin (*Tursiops truncatus*)

Entanglement data

A stranding based entanglement rate of 5.47% was reported for the Common Bottlenose Dolphin (*T. truncatus)* over the period of 1997-2009 in Florida, US based on data collected by various agencies (Adimey et al., 2014). This equates to an annual rate of 0.08% of the Florida coastal bottlenose dolphin population, after standardizing based on dolphin stock numbers from reports collated by NOAA (2020). The strandings were recorded throughout the coastal and inland Florida waters where it is known that high levels of fishing activity occur (Adimey et al., 2014). The rate reflects fishery gear interactions, and cases with other marine debris types were excluded from their analysis. The majority of the entanglements were caused by hook and line gear, and statistical testing showed that adult male dolphins were more strongly affected than females and juveniles (Adimey et al., 2014).

Potential exposure area

The Common Bottlenose Dolphin (*T. truncatus),* hereafter bottlenose dolphin, has a global distribution (Wells et al., 2019), occurring as both a coastal and an offshore ecotype. In the western North Atlantic, the coastal or inshore ecotype is found along the coast and in estuaries, bays and sounds, but there exists no firm boundary for where the range of the coastal ecotype stops and the offshore begins (Garrison et al., 2017).

A telemetry study from the coast of Georgia found that coastal dolphins were most abundant within 10 km from shore (Balmer et al., 2018). Others have suggested that bottlenose dolphins encountered more than approximately 23 km from the coastline can be considered the offshore type (Breen et al., 2016), while surveys from southeast Florida found that the offshore ecotype mainly occurred >34 km from shore (Herzing and Elliser, 2016). Similarly, a study on tissue samples suggested that dolphins from 34 km and beyond were the offshore ecotype, while dolphins encountered out to 7.5 km from shore were of the coastal ecotype (Torres et al., 2003). An unambiguous pattern was not established for the area between the two boundaries, and it is likely that the two ecotypes overlap around the continental shelf in the southern part of their range in the western Atlantic (Vollmer and Rosel, 2013).

The current entanglement study does not specify that no offshore dolphins were recovered stranded in Florida, but there are several reasons to believe that the entanglement data at large relates to the coastal ecotype. Offshore dolphins may also strand along the coast of Florida (Wells et al., 1999), but it is considered more rare that animals ranging over the continental shelf and beyond strands along the coast (Vollmer and Rosel, 2013). In situ experiments with dolphin carcasses carrying entanglements have shown that increasing distance from shore substantially decreases the likelihood of the carcass to strand (Peltier et al 2012). More specifically, Perrin et al. (2011) conducted genetic analyses of strandings and suggested that a carcass of the coastal ecotype was about 50 times more likely to strand than the offshore. An exclusion of the wide range of the offshore ecotype can further be supported by the current entanglements being linked to fishery activities in the coastal and inland waters of Florida through the debris types that was found on the stranded dolphins (Adimey et al., 2014).

Several studies indicate that there exist distinct groupings of dolphins exhibiting multi-generational site-fidelity to their communities along the Florida coast and maintaining small home ranges (Balmer et al., 2018, Urian et al., 2009, Defran et al., 1999). Although genetic analyses also support distinct geographical stocks, the coastal ecotype is found along the US coastline without physical barriers to dispersal (Rosel et al., 2009). As such, stocks of seasonally migrating coastal dolphins may also be found in Florida waters (Balmer et al., 2018) and strand there. However, the range of the migrating stocks can be excluded, as the majority of the entanglements were hook and line gear wrapped around dolphin appendages used for propulsion and maneuvering, such as the flukes (Adimey et al., 2014), which would likely hamper the mobility and migration potential of the individual. Owing to this and the overlap with fishery activities within the state, the potential exposure area can be limited to Florida. Taking into account the suggested cutoffs in the distribution of the ecotypes, a range of 30 km from shore extending all around the state of Florida is suggested to match the current entanglement data (Figure 10).

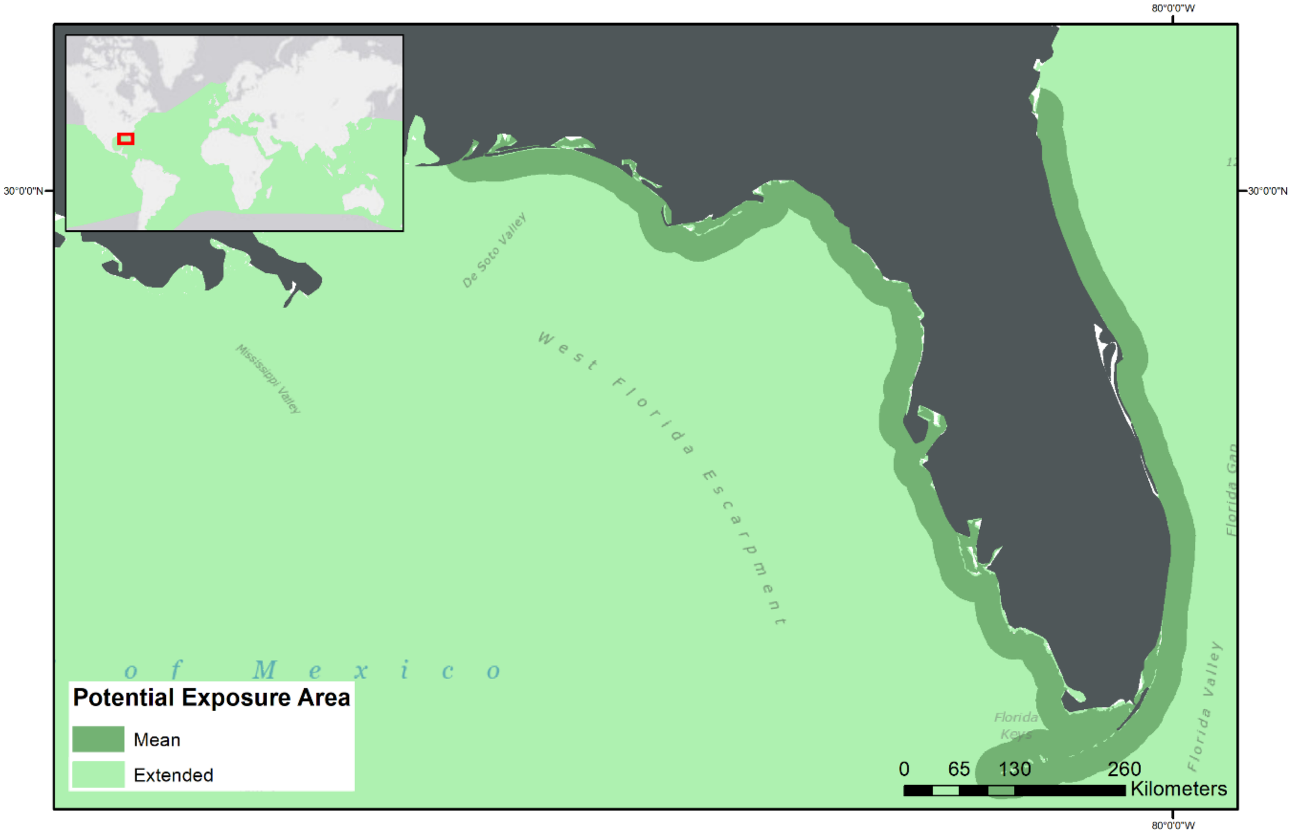


Figure 10: Mean potential exposure area for the Common Bottlenose Dolphin, as indicated by legend. Extent of the extended potential exposure area, which is the species IUCN range, is shown in the inset map.

4.2.7 Common Minke Whale (*Balaenoptera acutorostrata*)

Entanglement data

The entanglement rate for the Common Minke Whale (*B. acutorostrata)* over the period 2005-2010 was given as 9,1% based on stranding data from the UK Cetacean Strandings Investigation Programme (Werner et al., 2016), but could not be standardized to a population-wide annual rate owing to a low sample size and lack of population estimates. Representative entanglement rates are difficult to obtain for this population as most of the stranded whales are not investigated postmortem. However, 16 out of 30 determined mortalities between 1990-2010 in Scotland have been attributed to entanglement (Northridge et al., 2010), and 3 out of 4 in 2015 alone were caused by fishing gear entanglements (Deaville et al., 2016). This warrants a consideration of this species as sensitive to macroplastic debris entanglements. Owing to this, the potential exposure area was delineated for future modelling purposes only, as the current entanglement rate was excluded from the subset in this thesis.

Most of the strandings were registered off the coast of Scotland (Deaville et al., 2011), where entanglements appear to be the leading cause of death for the species (Northridge et al., 2010). The remainder were spread out over England, Wales, Northern Ireland and Isle of Man. The entanglements are most frequently caused by ropes, lines and other gear originating from creel fisheries (Deaville et al., 2016).

Potential exposure area

The Common Minke Whale (*B. acutorostrata*), hereafter minke whale, is found in coastal and offshore areas in all oceans of the world (Cooke, 2018), albeit most commonly in cooler waters of shelf areas of less than 200 m depth (Hammond et al., 2006, Breen et al., 2016, Northridge et al., 1995). Different subspecies of the minke whale are acknowledged, whereas *B. a. acutorostrata* is the type found in the North Atlantic, believed to range from the Barents Sea and to the west African continental shelf (van Waerebeek et al., 1999). In the North Atlantic, the subspecies is further separated into several management stocks by the International Whaling Commission (IWC) (VKM et al., 2019). A separation between a central and an eastern North Atlantic stock is often applied, although there are doubts about the biological relevance of this boundary as the stocks are believed to overlap on their feeding grounds (Skaug et al., 2004, Anderwald et al., 2011).

Delineating a relevant foraging range for this cosmopolitan and highly mobile mammal with a likely ephemeral and not well established winter range (Macleod et al., 2004, Anderwald et al., 2011) and unknown breeding ground locations (VKM et al., 2019, Risch et al., 2019) is not feasible. Minke whales range widely across the North Atlantic and due to this migratory behavior there appears to be limited genetic differentiation separating them into distinct populations that can be assigned to geographical areas (Anderwald et al., 2011). However, the potential exposure area for the current entanglement rate can be narrowed down to the waters around the UK where the strandings are reported (Deaville et al., 2011). The entanglement rate is based on stranding data with entanglements as the cause of death, which implies that the injuries are more acute, rather than chronic entanglements that leave scarring but do not cause mortality. It has also been suggested that entanglements more frequently lead to death for smaller cetaceans such as minke whales (Dolman and Moore, 2017). Taking this into account, it is deemed less likely that the whales would carry entanglements through several seasons and strand in areas far away from the point of entanglement.

The coastal and shelf waters around the UK make up important summer feeding grounds for the eastern Atlantic population (Breen et al., 2016, Robinson et al., 2009), but sightings are reported year-round (Macleod et al., 2004). The whales are most frequently observed at water depths from 20-50 m on their feeding grounds in the coastal waters of northeast Scotland (Robinson et al., 2009), and distances to shore during summer has been seen to vary from 1-15.6 km (Risch et al., 2019). However, they are found further from the coast outside the summer (Risch et al., 2014), and known to frequent offshore areas such as Rockall Bank west of Scotland (Hammond et al., 2006). However, the entanglements are suggested to be linked to gear from static fisheries in coastal waters (<100 m depth) such as ropes, pots and gillnets (Deaville et al., 2016, Kot et al., 2012), favoring a focus on waters within the shelf areas. As such, predicted monthly distribution maps for minke whales around the UK are used as the basis for the potential exposure area (Waggitt et al 2020) along with the sources to abundance information presented above. The potential exposure area is mainly set within the 200m contour, but extends further in areas of known occurrence and excludes areas where they are not commonly observed (Figure 11).

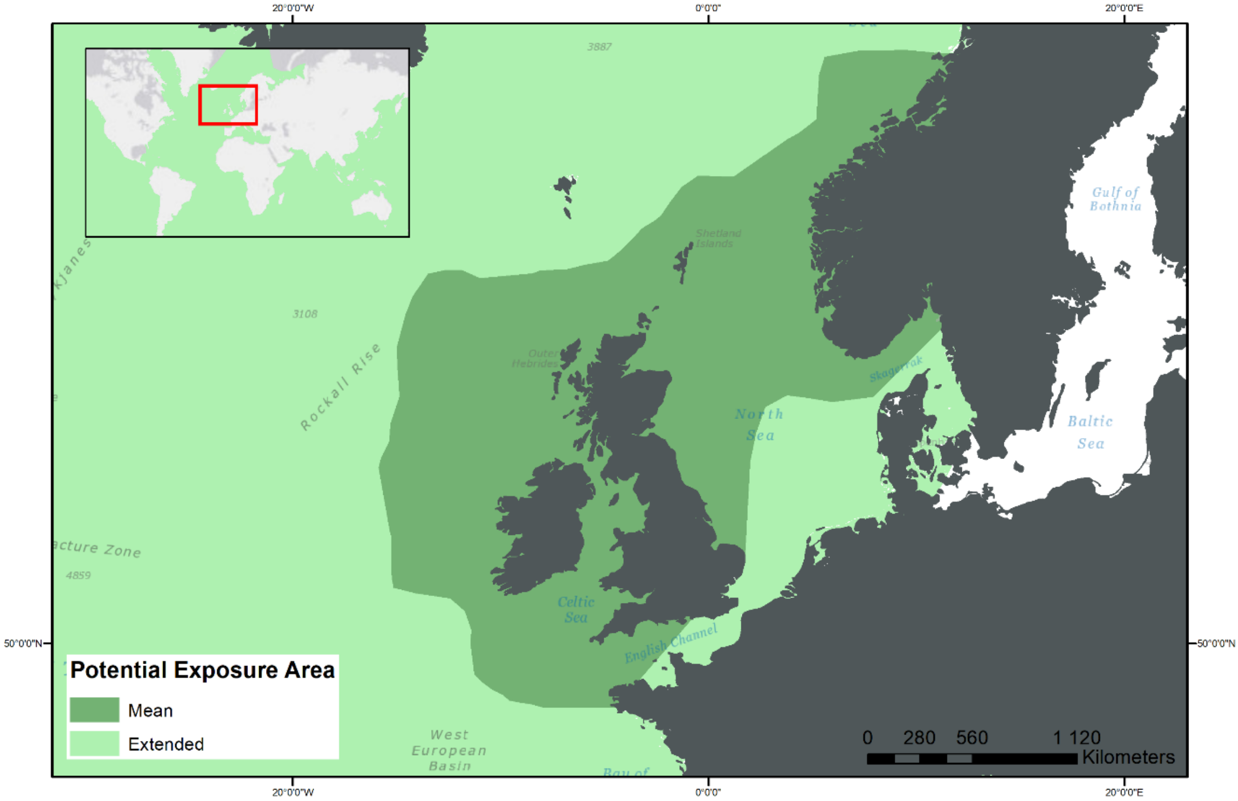


Figure 11: Mean potential exposure area for the Common Minke Whale, as indicated by legend. Extent of the extended potential exposure area, which is the species IUCN range, is shown in the inset map.

4.2.8 Florida Manatee (*Trichechus manatus latirostris*)

Entanglement data

A stranding based entanglement rate of 7.66% was reported for the Florida Manatee (*T. m. latirostris*) over the period of 1997-2009 in Florida, US based on data collected by various agencies (Adimey et al., 2014). This equates to an annual rate of 0.46% of the Florida population, after standardizing based on a statewide abundance estimate (Martin et al., 2015). The strandings were recorded throughout the coastal and inland Florida waters, where it is known that high levels of fishing activity occurs (Adimey et al., 2014). The rate is based exclusively on fishery gear interactions, and cases with other marine debris types were excluded from their analysis. Hook and line, followed by trap pot gear was responsible for the majority of the entanglements (Adimey et al., 2014).

Potential exposure area

The Florida Manatee (*T. m. latirostris),* hereafter manatee, is a subspecies of the West Indian Manatee distributed along the coast of the southeastern United States, but mainly found in the coastal nearshore waters of Florida, including freshwater and estuarine ecosystems (Flamm et al., 2005). Although parts of the population migrate out of Florida and to nearby states during summer (Deutsch et al., 2008), the spatial overlap and gear specific link between the entanglements and local fishing activity (Adimey et al., 2014) allows for a consideration of the Florida range as the potential exposure area.

The manatees aggregate in warm-water sites such as power plant thermal outfalls and artesian springs in Southern Florida during winter (Laist and Reynolds, 2005), and they require access to inland drinking water (Langtimm et al., 2011). Despite their utilization of these non-marine areas, they are exposed to marine debris when foraging on seagrass patches along the coast and when using the coastline to move between habitats and aggregation sites (Flamm et al., 2005). Owing to differences in habitat requirements, the species exhibit distinct seasonal distributions (Langtimm et al., 2011, Semeyn et al., 2011), but as the entanglement data is averaged over all seasons, the range applied here will reflect a general distribution accounting for the whole year. In addition, the subspecies can be further separated into subpopulations that share distribution patterns (Deutsch et al., 2008), but as the entanglement data is given on the level of Florida state, this will not be considered.

Tracking studies of manatees from Tampa Bay show that they are capable of travelling long distances of over 100 km, e.g. from the west coast of Florida to the east (Weigle et al., 2001), but there is paucity in information of exactly how far they range from the coastline. This is likely owing to the fact that this subspecies inhabit only the shallow waters of Florida, seldomly found at depths beyond 3.7 m (Edwards et al., 2016b, Martin et al., 2015). However, as this implies a narrow range around the coast which may be influenced by tidal fluctuations, wind and small-scale variations in bathymetry, a range extending to 5 m depth will be used to encompass the potential exposure area (Figure 12). This is also within the range of the species in general, which is most commonly found at depths between 2-6 m, but occasionally at 10 m (Castelblanco-Martínez et al., 2009, Lefebvre et al., 2000, Olivera-Gómez and Mellink, 2005).

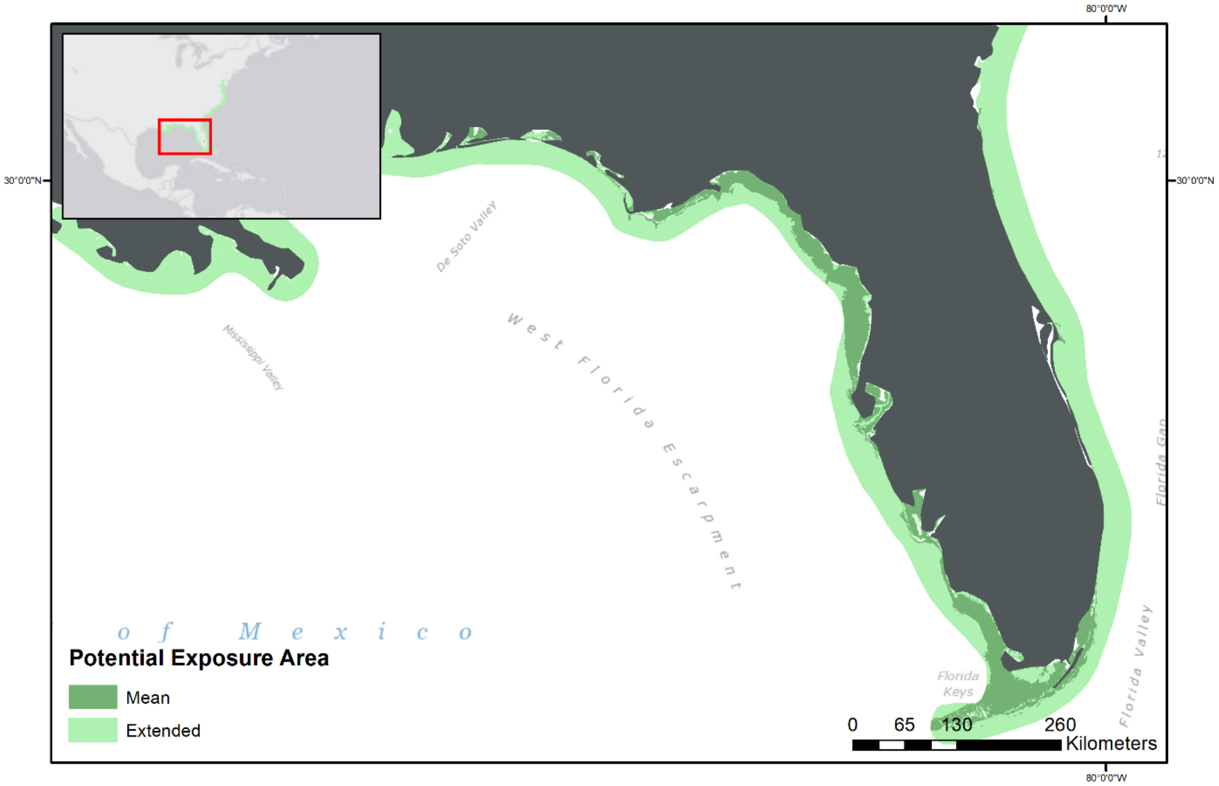


Figure 12: Mean potential exposure area for the Florida Manatee, as indicated by legend. Extent of the extended potential exposure area, which is a modified subspecies IUCN range, is shown in the inset map.

4.2.9 Grey Seal (Halichoerus grypus)

Entanglement data

An annual entanglement rate of 4.3% was estimated for the Grey Seal (*H. grypus)* based on sightings and a photo identification catalogue from a haul-out site in Cornwall, UK from the period 2004-2008 (Allen et al., 2012). The authors suggested that severe injuries were associated with 64% of all entanglement cases observed and that this could indicate a potential high rate of mortality (Allen et al., 2012). In the cases where entangling material could be identified, it was found to be mainly fisheries related. This can be linked to the intensity of fishing effort in the area, resulting in large quantities of fishing debris found around the coast of the study site (Allen et al., 2012).

Potential exposure area

The Grey Seal (*H. grypus)* is found in the continental shelf regions of the North Atlantic and can be separated into a North East and a North West population (Bowen, 2016). The grey seals are year-round residents at haul-out sites in Cornwall, although higher numbers can be found during the moulting and breeding season (Leeney et al., 2010). During these seasons, they act as central place foragers, while they spend long periods at sea for the remainder of the year (Vincent et al., 2017). Although grey seals are not long-distance migrants (Bowen, 2016), telemetry studies have found that seals from Cornwall move back and forth across national and international borders, such as to Wales and France (Sayer et al., 2019, Vincent et al., 2005). This implies that the seals in Cornwall are part of a metapopulation spanning the Celtic sea (Leeney et al., 2010). However, they are also found to exhibit site fidelity to breeding sites (Pomeroy et al., 1994), foraging regions and haul-out sites (Russell et al., 2013, Vincent et al., 2005). Studies from other regions have found that females often feed closer to the haul-out sites than males (Breed et al., 2009) and that juveniles in general move over large areas (Sjöberg et al., 1995).

Although grey seals may travel several hundred kilometers to forage (Mcconnell et al., 1999), their movements are often restricted to smaller areas close to haul-out sites (Oksanen et al., 2014). In North East England, they have been found to travel on average about 40 km from the haul-out site on each return-trip (Mcconnell et al., 1999), while individuals tracked in the Baltic sea used an area within a 50 km radius of their site for 75% of the time (Sjöberg and Ball, 2000). This is in agreement with newer studies from Celtic and Irish waters where it has been found that grey seals generally forage within 50 km of their natal haul-out sites or colonies (Gosch et al., 2019, Carter et al., 2017, Cronin et al., 2012).

As a species that favor shallow water areas, the grey seals inevitably overlap with coastal fisheries (Oksanen et al., 2014), and particularly inshore passive fisheries consisting of nets and lines in Irish waters (Cronin et al., 2016). A study on marine debris around the UK found that the Western English Channel had the highest mean abundance of debris (Nelms et al., 2017), and the waters around Cornwall are known fishery hotspots (Leeney et al., 2010). This makes the local population of seals potentially exposed to large amounts of both fishing related debris and operational fishing gear, and it favors a coupling of the estimated entanglement rate to a local foraging range. The entanglement study does not specify from which haul-out site on Cornwall the rates were obtained, but it is likely that the seals make use of several haul-out sites (Gosch et al., 2019) distributed all around Cornwall and the nearby Isles of Scilly (Leeney et al., 2010). As such, Cornwall including the Isles of Scilly will be included in the potential exposure area, which extends 50 km from all sites along the coast as well as around the Isles of Scilly (Figure 13).

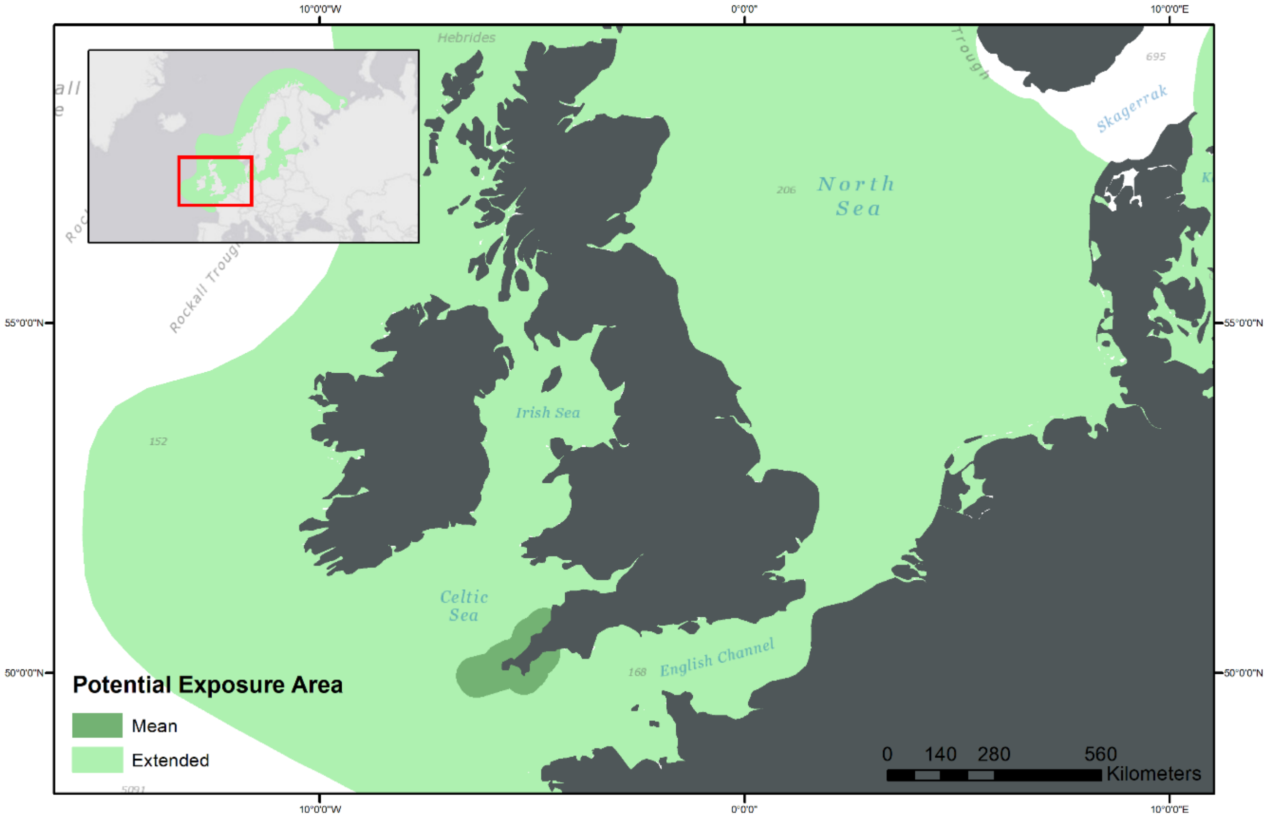


Figure 13: Mean potential exposure area for the Grey Seal, as indicated by legend. Extent of the extended potential exposure area, which is a modified IUCN range, is shown in the inset map.

4.2.10 North Atlantic Right Whale (Eubalaena glacialis)

Entanglement data

Based on observations of entangled whales collected in the North Atlantic Right Whale Identification Database and population estimates from 2017 through 2019 (Pettis et al., 2020), it was derived that 1.88 % of the population annually gets entangled, of which 1.4% are injuries and 0.47% are confirmed entanglement mortalities. The entanglements have been registered along the East coast of North America, from Florida and up to Newfoundland (Pettis et al., 2020).

Potential exposure area

The North Atlantic Right Whale (*E. glacialis*), hereafter NARW, inhabits the coastal waters of the eastern coast of the United States and Canada (Hunt et al., 2015), ranging from Florida to southeast Labrador (Jacobsen et al., 2004). The species is highly transitory and moves frequently between habitats covering vast areas within a season (Brillant et al., 2015, Baumgartner et al., 2017). Studies indicate that the western Atlantic population constitute one interbreeding population, but there is a paucity in knowledge about overlaps with a potential population in the Eastern Atlantic (Cole et al., 2013). There are currently no regular sightings in this part of their historical range (Knowlton et al., 1992, Harcourt et al., 2019), making the existence of a remnant north eastern Atlantic population questionable (Cooke, 2018). Owing to this, the distribution map collated by the IUCN represents where the species historically occurred, and may occur based on oceanography, but does not reflect the current documented distribution (Cooke, 2018). However, as two-way trans-Atlantic migrations have been observed (Jacobsen et al., 2004), the potential for whales to get entangled in the Eastern Atlantic and subsequently sighted in the Western Atlantic should be considered. This is relevant for a large mammal like the NARW, which may be capable of surviving with the extra weight and injuries from entanglements for years (van der Hoop et al., 2017). Although entanglements may also seriously impede the foraging ability of NARW, as migrating capital breeders, this species is adapted to going longer periods with limited food intake (Lysiak et al., 2018). However, the vast majority of the entanglements occur with fishing gear linked to extensive fisheries in their foraging areas off the North American east coast (Good, 2008). Moreover, this small and endangered population is well monitored (Harcourt et al., 2019), and the entanglement records contain the locations of where the individually identified whales were first seen entangled (Pettis et al., 2020). As such, omitting the eastern Atlantic region from the potential exposure area is reasonable.

The whales migrate seasonally from winter calving areas off Florida and Georgia to well-known feeding grounds found from Cape Cod Bay and up to the Gulf of St. Lawrence (Cooke, 2018, Pettis et al., 2020), utilizing the seaboard of the Mid-Atlantic region as their primary migratory corridor (Firestone et al., 2008). Coastal areas within 40-50 km of the shore have been suggested to make up the core calving habitat in the southeast, with a mean range from shore of around 17 km (Good, 2008, Gowan and Ortega-Ortiz, 2014). Telemetry data and visual observations have shown they stay well within 50 nmi (92.6 km) of the coast in the Mid-Atlantic region, and sightings beyond 30 nmi (55.56 km) were generally in the northern part of the study area. Although they are found to be more mobile in the Northern part of their range, most tracked locations have been along bank edges, in basins or along the continental shelf (Mate et al., 1997)

The geographic locations of the entanglement events can usually not be determined with accuracy (Lysiak et al., 2018), but the records of sighted entangled whales indicate that they can get entangled throughout their North American range (Knowlton et al., 1992), likely occurring along the coast (Good, 2008). As such, the range of the western Atlantic population assumed for the purpose of this thesis extends from Florida to Newfoundland. Although historical sightings around Newfoundland are sparse compared to the rest of the coast, the Gulf of St. Lawrence has more recently been suggested as an important NARW area that is less frequently surveyed (Brillant et al., 2015, Harcourt et al., 2019). As whales are not frequently observed in the Gulf of Mexico, likely due to a thermal barrier (Good, 2008), the west coast of Florida is excluded from the potential exposure area.

Critical habitat areas assigned to this species (NOAA, 2016) and the abundance of point observations from OBIS (2020) Seamap coincide with bathymetry of 0-200 m along the North American coastline. This is also supported by the general pattern of their distribution further from the coast along a gradient from south to north, as outlined above. As such, the potential exposure area extends from the North American coast and to waters within 200 m depth (Figure 14). The southern boundary is determined by the extent of the known breeding ground area, while the northern boundary is set where OBIS (2020) observation points become infrequent and no known aggregations occur.

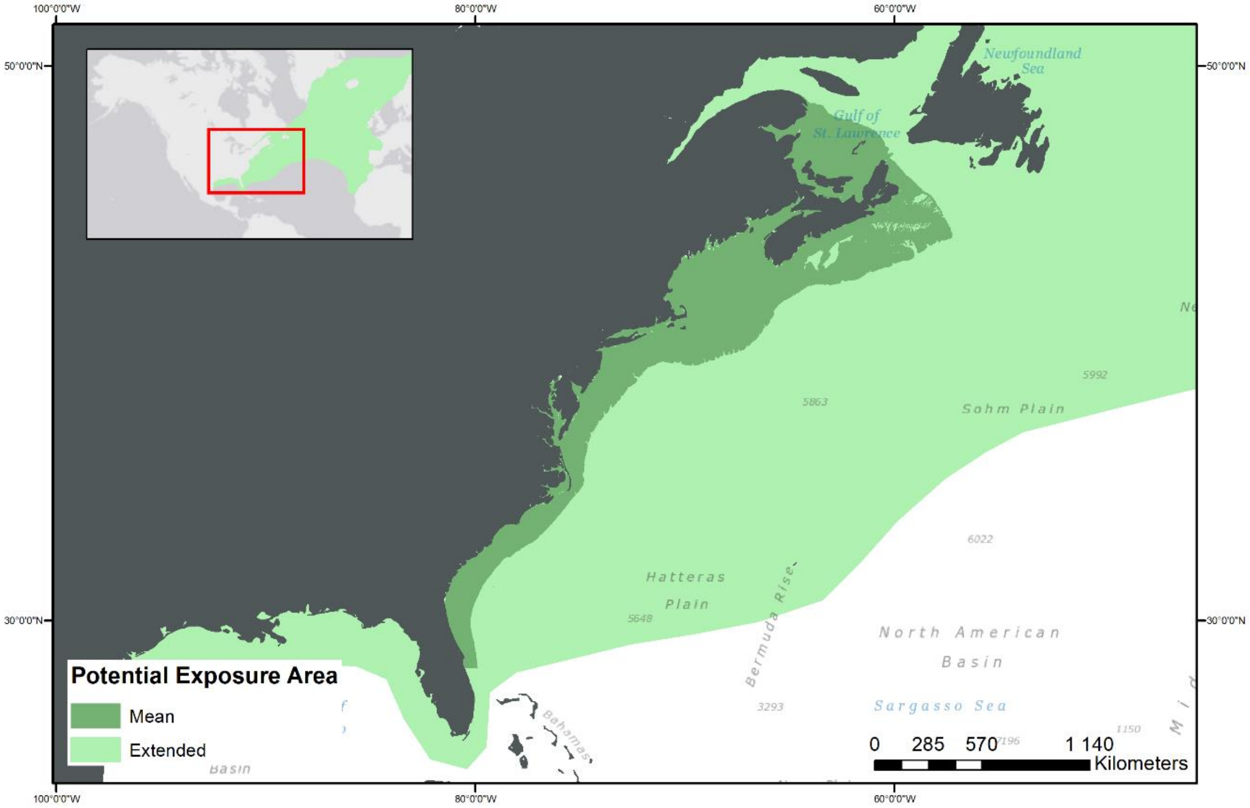


Figure 14: Mean potential exposure area for the North Atlantic Right Whale, as indicated by legend. Extent of the extended potential exposure area, which is the species full IUCN range, is shown in the inset map.

4.2.11 Steller Sea Lion (*Eumetopias jubatus monteriensis*)

Entanglement data

An annual entanglement rate of 0.26% was reported for the Steller Sea Lion (*E. jubatus)* from 2001-2007 in Southeast Alaska and Northern British Columbia (Raum-Suryan et al., 2009). The entanglement rate is based on surveys from rookeries and haul-out sites conducted from June through August, where the total number of individuals observed as well as a population estimate was taken into account (Raum-Suryan et al., 2009). The most common form of entanglement was packing bands encircling the neck of the seals, but in most cases it was so deeply embedded that the material could not be identified (Raum-Suryan et al., 2009).

Potential exposure area

There are two recognized subspecies of Steller Sea Lions, namely the Western (*E. j. jubatus) and* Loughlin's (*E. j. monteriensis) (Gelatt and Sweeney, 2016a). The latter is often referred to as the Eastern Distinct Population Segment (Gelatt and Sweeney, 2016b), and* the current entanglement rate is obtained within the IUCN range of this population. The subspecies breeds along the west coast of North America, ranging from California and through Southeast Alaska to 144° W longitude (Cape Suckling, Alaska) (Loughlin, 1997).

Like other otariids, Steller Sea Lions are central place foragers who generally display site-fidelity by returning to their terrestrial birth site to breed (Loughlin, 1997, Gelatt and Sweeney, 2016a). As such, interchange between the Eastern and the Western stock has been thought to be limited (Raum-Suryan et al., 2002), but the presence of a mixing zone facilitates movements of both females (Pitcher et al., 2007) and males between the stocks (Jemison et al., 2013, Hastings et al., 2019). The two subspecies are also known to overlap outside the breeding season (Loughlin, 1997). However, as the entanglement data is mainly collected within the breeding season, the potential exposure area will be limited to the present range of the Eastern population. The range south of British Columbia may also be excluded from the potential exposure area based on the same premise, and because only a small minority of the population is found there (Gelatt and Sweeney, 2016b).

During the breeding season, females need to nurse their pups and maintain a restricted foraging range (Gelatt and Sweeney, 2016a). Both females and pups are found to largely remain within 17 km of their rookeries (Rehberg et al., 2009, Loughlin, 2009). Although there is a lack of studies on the movements of adult males (Boor and Small 2012), they are known to be strongly tied to the rookeries during the breeding season (Jemison et al., 2013, Cornick et al., 2006) when they fast and maintain territories (Womble et al., 2009). Juveniles are also seen to use the nearshore areas (< 15 km) around haul-out sites most frequently when conducting round trips (Raum-Suryan et al., 2004). However, juveniles may also undertake longer trips and range as far as >2000 km away from their natal rookeries (Jemison et al., 2018). As the entanglement data also spans the whole of August when the breeding season has ended and seals are less tied to the rookeries, a range of 15 km is too narrow to be representative of their potential exposure area.

Jemison et al. (2018) analyzed >30 000 sightings collected over 14 years of sea lions from Alaskan rookeries and provided the mean distance observed from rookeries for males and females spanning all age classes. Combing the age classes and sexes and only considering the Eastern population, a mean distance of ca 123 km is obtained for the breeding season. This is considerably larger than the suggested range of breeding adults (Rehberg et al., 2009) and smaller than the juveniles' dispersal potential (Jemison et al., 2018, Raum-Suryan et al., 2002). However, as all age and sex classes were observed entangled, a consideration of the population as a whole is deemed necessary. As such, the potential exposure area is set to a 123 km radius around each rookery and known haul-out site in Southeast Alaska and British Columbia (Figure 15)

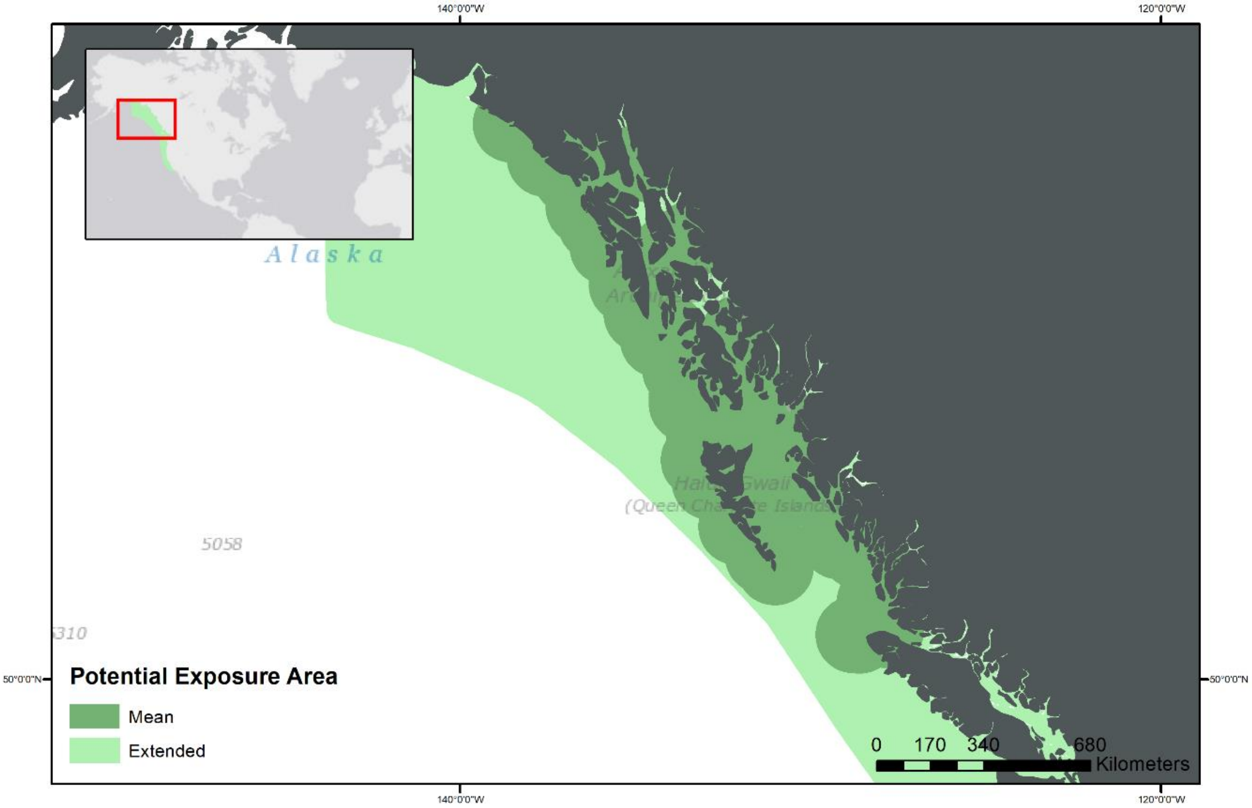


Figure 15: Mean potential exposure area for the Steller Sea Lion, as indicated by legend. Extent of the extended potential exposure area, which is the subspecies IUCN range, is shown in the inset map.

4.3 Robustness of potential exposure areas

Although a general approach was followed, potential exposure areas may be delineated in several ways for the different species populations, as the process is directly dependent on what is available in literature data. The potential exposure areas were defined on the basis of foraging ranges and abundance data estimated using different methods, which in turn will be associated with different uncertainties. Moreover, to what extent it was possible to obtain robust estimates of the mean ranges of specific populations rather than the general species range was highly variable. As such, the delineated areas may be over- or underestimates in terms of representing the actual exposure areas of a population linked to an entanglement rate. This yields differences in the level of confidence in the match between a given site-specific entanglement rate and the corresponding delineated potential exposure area (Table 2).

Table 2: Level of confidence in the match between entangled population and delineated mean potential exposure area.

Species	Area delineation basis[A]	Confidence level[B]
Northern Gannet (*M. bassanus*) Breeding colony	Region specific tracking	Moderate
Northern Gannet (*M. bassanus*) Non-breeding sites	Species general tracking	Low
Common Guillemot (*U. aalge*)	Colony specific observations	Moderate
Northern Fulmar (*F. glacialis*)	Species general tracking	Low
Antarctic fur seal (*A. gazella*)	Colony specific tracking	Moderate
Australian Fur Seal (A. p. doriferus)	Colony specific tracking	High
Common bottlenose dolphin (*T. truncatus*)	Region specific tracking	Moderate
Florida Manatee (*T. m. latirostris*)	Region specific tracking	High
Grey seal (*H. grypus*)	Region specific tracking	Moderate
North Atlantic right whale (*E. glacialis*)	Region specific tracking	High
Steller Sea Lion *(E. j. monteriensis*)	Colony specific tracking	High

[A]: The table depicts only the main basis for delineating the mean potential exposure areas.

[B]: See Appendix 1 for criteria applied in assigning confidence levels.

In general, high resolution surveys of wide-ranging marine mammals are rare, as the practice of observing animals at sea is costly and time consuming. However, considering that the objective of this work was to identify mean ranges or likely areas of high use for the majority of the population, a level of confidence in the coupling could be achieved. The Common Bottlenose Dolphin (*T. truncatus*) was assigned a moderate confidence level owing to the existence of two ecotypes with different ranges, yet potential for spatial overlap. For small and well monitored populations such as the North Atlantic Right Whale (*E. glacialis*), tracking, visual and acoustic records were available, which warrants a high confidence in the potential exposure area despite the large mobility potential of this species. Moreover, several tracking studies have been conducted for all the pinniped species from the specific colonies, which enables a high confidence in the delineated areas. However, paucity in tracking of parts of the population, specifically adult males, lowers the confidence for the range of the Antarctic Fur Seal (*A. gazella*).

Although there exists temporally and spatially matching tracking studies for the foraging range of the breeding Northern Gannet (*M. bassanus)*, the confidence in the estimate is lowered by the uncertainty linked to the potential difference in extent travelled for nesting material collection as opposed to foraging. A low confidence was assigned to the mean area delineated for the Northern Fulmar (*F. glacialis*) as the range was obtained from species general tracking and the ranges of fulmars are known to be highly variable from colony to colony (Mallory et al., 2012).

4.4 Entanglement rates coupled with plastic debris exposure

4.4.1 Sensitivity of species

Entanglement rates were linked to spatially specific mean plastic debris exposure concentrations in order to derive the modelled plastic concentrations at which 5% of the species population is affected (EC5) by macroplastic debris annually. Among the species populations in the current entanglement dataset, the Antarctic Fur Seal *(A. gazella*) was predicted to be the most sensitive to macroplastic entanglement (Table 3). This can be explained by their habitat in the Southern Ocean having the lowest estimated plastic concentrations. Conversely, The Australian Fur Seal (*A. p. doriferus*) population can be identified as the most tolerant species, and forages in an area with high estimated plastic concentrations. Moreover, this difference in sensitivity between the fur seal species is linked to the observed entanglement rate level of the Antarctic seal population being higher than that found for the Australian seals (Table 3). As such, the coupling of a higher entanglement rate to an area with lower mean estimated plastic concentrations predicts that the species is more prone to becoming entangled even when plastic debris is less abundant in its core habitat.

Table 3: Brief description of mean potential exposure areas with associated plastic concentrations and entanglement rates, used to derive each of the EC5's (modelled plastic concentration at which 5% of the population is entangled annually). Sorted from most sensitive (top) to least (bottom).

Species	Mean potential exposure area	Mean plastic conc. (g/km)	Entanglement rate (%)	EC5
A. gazella	South Georgia: 500 km buffer around Bird Island and Cumberland Bay	3,73E-01	0.05	4,34E+01
	South Orkney: 500 km buffer around Signy Island	2,87E-01	0.02	
H. grypus	Cornwall and Isles of Scilly: 50 km buffer around haul-out sites	8,84E+02	4.30	1,03E+03
E. j. monteriensis	SE. Alaska and N. British Columbia: 123 km buffer around haul-out sites	8,86E+01	0.26	1,70E+03
M. bassanus	Heligoland: 12,5 km buffer around breeding colony	9,93E+02	3.05	2,03E+03
	Gulf of Cadiz: 135 km buffer around survey area	2,24E+02	0.11	
	Cantabrian sea: 135 km buffer around survey area	1,58E+03	0.36	
	Mauritania: 135 km buffer around survey area	2,20E+02	20.16	
F. glacialis	Heligoland: 47.5 km buffer around breeding colony	1,03E+03	1.80	2,86E+03
E. glacialis	Atlantic coast of North America: Area with depth of 0-200 m	1,53E+03	1.88	4,06E+03
U. aalge	Heligoland: 20 km buffer around breeding colony	9,93E+02	1.05	4,73E+03
T. m. latirostris	Florida: Area with depth of 0-5 m around the peninsula	4,11E+03	0.46	4,47E+04
T. truncatus	Florida: 30 km buffer around the peninsula	4,93E+03	0.08	3,16E+05
A. p. doriferus	Southern Victoria: 200 km buffer around Seal Rocks Island	3,36E+03	0.014	1,20E+06

For most of the species populations, the EC5 is based on only one datapoint, as the entanglement rate(s) pertained one population for which a single potential exposure area could be delineated. However, two geographically distinct areas with associated plastic concentrations and entanglement rates were found for the Antarctic Fur Seal (*A. gazella*). Although estimated plastic debris concentrations are remarkably low in this region, it can be seen that the entanglement rate is higher in the area of higher concentrations (Table 3).

For the Northern Gannet *(M. bassanus*), the two separate EC5's modelled to conserve differences in exposure and sensitivity related to distinct breeding behavior, were combined in a final weighted EC5 for the species. When investigating all areas separately, the mean plastic concentrations at which 5% of the population is precited to be entangled are highly variable across sites, spanning three orders of magnitude (Figure 16).

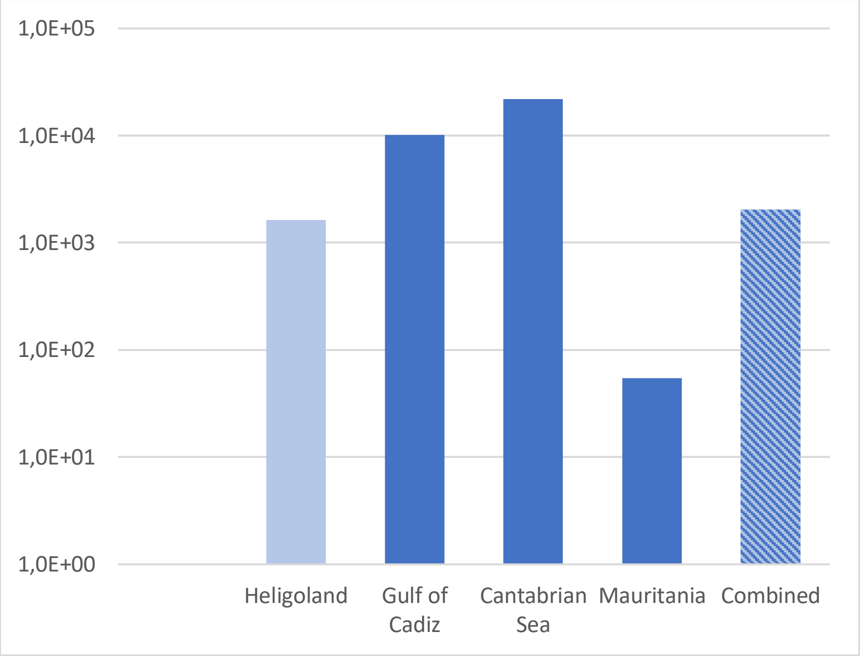


Figure 16: Estimated plastic concentrations (g/km^2) at the breeding colony (light blue), the wintering sites (blue) and the weighted average (striped) for the Northern Gannet when 5% of the population is affected by entanglement annually (EC5).

As this level of detail was not available for the other species populations in the current work, the Northern Gannet (*M. bassanus)* serves as an example of how different the EC5 for a species may be depending on which sampling locations, in addition to parts of the species' annual cycle, that are included. This is particularly evident when considering the wintering site Mauritania, where birds were registered entangled at much higher rates than any other site in the current dataset, while the corresponding estimated plastic concentration is lower than that of the other wintering site (Figure 16). This is at odds with the linear approach assumed for the modelling, and highlights the need for obtaining additional datapoints in order to better model the concentration-response relationship to be used for the species sensitivity distribution (SSD) curve.

Differences in entanglement prevalence for a population throughout their annual cycle may however be accounted for, albeit not distinguishable, for the species populations where surveys were conducted year-round and an average rate was given, such as for three of the pinniped species (Table 1).

4.4.2 Influence of modelling choices

Owing to the potential for bias and ambiguity in population range delineations, it is useful to test to what degree applying differently sized exposure areas affects the resulting EC5 for a given species. For the three seabird species populations and the Antarctic Fur Seal, a maximum range was obtained in addition to the mean potential exposure area, while the full IUCN ranges or modified IUCN subspecies ranges were applied as the extended range for the remainder of the species (Appendix 2).

Both the extent of the exposure area and the choice of summary statistics for calculating the area-specific plastic concentrations influence the resulting plastic concentration coupled to the entanglement rate. Although caution needs to be exerted because the increase in the potential exposure area from the mean to the extended range was not uniform across species, a comparison of the rankings was done in order to gain insight into how sensitive the model is to the way the area is defined. The EC5 values varied by six orders of magnitude depending on the species population and the modelling choices (Figure 17).

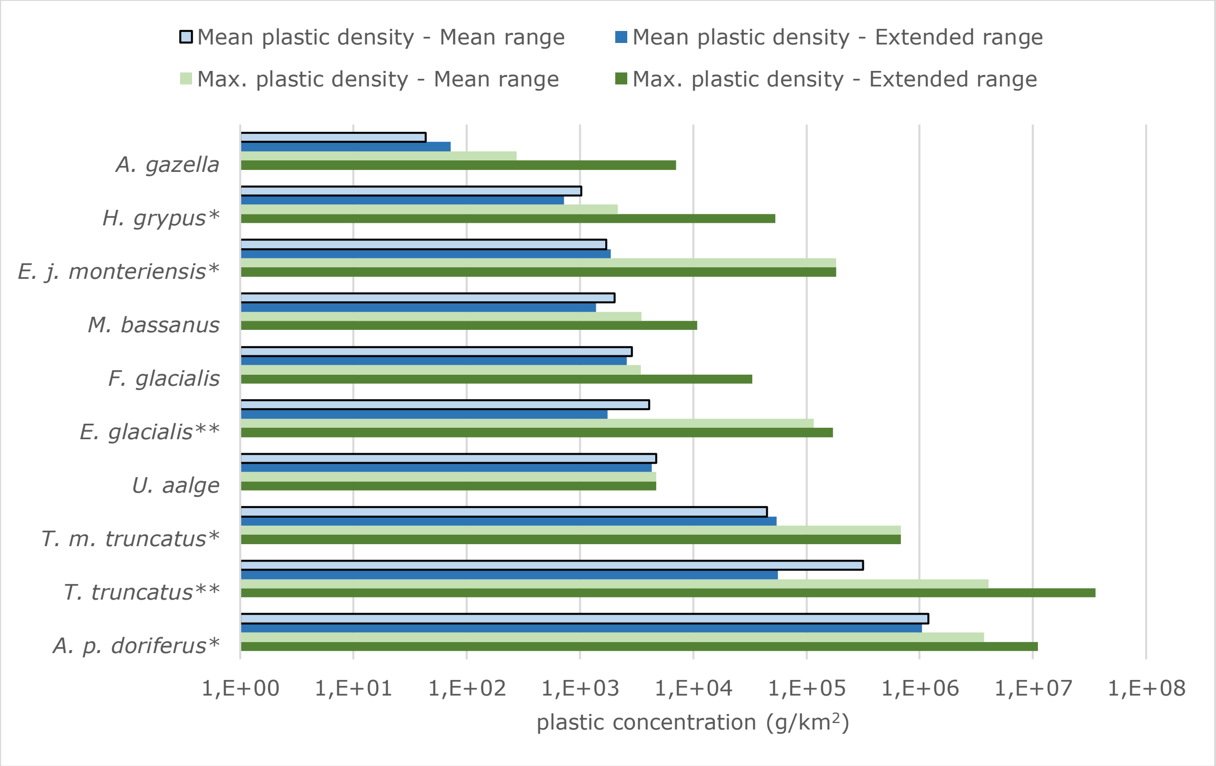


Figure 17: Species sensitivities based on the environmental concentration of plastic debris (g/km^2) at which 5% of a population is predicted entangled (EC5), under four modelling combinations, as indicated by color (see legend). Species are ordered from most sensitive (top) to least (bottom) according to the EC5 for the mean potential exposure area using mean plastic concentrations.

* Modified IUCN subrange applied as extended range; ** Full IUCN range applied as extended range.

Based on the current entanglement data, the Antarctic Fur Seal *(A. gazella*) comes out as the most sensitive species in all but one combinations of modelling choices, while the Australian Fur Seal (*A. p. doriferus*) population is predicted to be the least sensitive in two out of the four rankings (See Appendix 3 for detailed EC5's).

The Common Guillemot (*U. aalge*) has the most stable EC5 across modelling choices (Figure 17) which can be attributed to a small range and the marginal increase in the size of the potential exposure area from the mean to the extended range. Indeed, the smallest ranges delineated in this work (≤ 20 km radius around a single point location) were not influenced by the choice of plastic concentration calculation, as they were found to only encompass one grid-cell assigned one value. As such, variations on a small spatial scale are negligible when applying the present plastic concentration model, although there may be slight variations depending on the geographical location of the range, as the cell sizes are in degrees and not km.

Overall, it is apparent that for most other species populations, the use of the maximum value as the plastic concentration in combination with the extended range, regardless of whether it is population specific or IUCN derived, decreases the predicted sensitivity. For example, when coupling the entanglement rates with the complete IUCN distribution of cosmopolitan species like the Common Bottlenose Dolphin (*T. truncatus*), the maximum plastic concentration will naturally be high, and concomitantly the sensitivity of the species is found to be low. In addition, the mean plastic concentration estimated for such a vast range is also less meaningful in spatially explicit assessments as it will be close to a global mean. However, the dolphin population also appears among the least sensitive species when applying the delineated mean potential exposure area (Table 3). A higher EC5 will be obtained when the entanglement rate is relatively low while the plastic concentration within the area is high, which can imply that the species is in fact less sensitive to plastic debris entanglement due to e.g. avoidance behavior. Owing to a limited dataset, it is however not possible to ascertain any relationship as the observed sensitivities may also be a consequence of an underestimated entanglement rate, or a mismatch in the estimated exposure to plastic debris.

5 Conclusions and Future Considerations

Quantifying the effects of marine plastic debris in general, and delineating exposure areas for marine species in particular, remains a complex task, as it involves the collection and integration of several layers of information on multiple species, all associated with different life history traits that affect their exposure to plastic debris (Compa et al., 2019). Mean potential exposure areas were delineated for the purpose of representing where a population linked to an entanglement rate is most likely to encounter plastic debris. In the current work, equal weighing on all areas within the range was assumed. However, as time spent in an area is an important determinant of exposure, assigning different weighing to zones within delineated areas is suggested for future modelling. This could more adequately represent the areas where a population is more likely to interact with plastic debris on a finer level, while avoiding the exclusion of areas of less use that they pass on their way to specific high-use foraging locations. This is particularly relevant for seabirds who will not interact with the sea surface for great lengths of their foraging trips. However, this type of modelling is challenging in practice as many species are opportunistic and less predictable in their foraging patterns, and areas where they forage may change annually and seasonally.

The actual exposure areas of the species populations represented in this work can potentially be larger or smaller than the mean extents defined, as it is not certain to what degree the mobility and dispersal potential of an animal is hampered as a result of an entanglement. It may be assumed that entanglements restrict the movement of the individual, implying that they are more likely to get entangled closer to maximum where they are observed. This was the general assumption followed in this work, and as such an additional argument for placing the focus on mean over maximum ranges. However, to what extent mobility is limited remains difficult to quantify as it will be highly variable depending on the individual, the species and the severity of the entanglement.

In deriving the EC5 for the species in this work, the distribution and abundance of macroplastic throughout the water column is not accounted for as the current dataset only estimates surface water plastic debris concentrations. Large discrepancies between estimated amounts of emitted plastic waste (Jambeck et al., 2015) and that estimated for the surface waters (Eriksen et al., 2014) indicates that most of the mismanaged plastic ending up at sea sinks in the water column and eventually sediments (Fazey and Ryan, 2016). As such, the actual plastic concentrations that the species in this work are exposed to are likely higher than modelled, and their predicted sensitivities may be overestimated when the plastic concentrations are underestimated. This could perhaps be the case for the Antarctic Fur Seal, which was found to be the most sensitive species when comparing the preliminary EC5's. It may however be argued that the sensitives are in absolute terms unlikely to be overestimated, when considering that the entanglement rates can only represent a snapshot of the prevalence of plastic debris entanglements. However, when plastic debris models can account for the heterogenous distribution throughout the water column, species can be categorized into what habitat they predominantly use (e.g. pelagic or demersal) to get a more accurate representation of what types of plastic debris they are exposed to.

Moreover, no discrimination in the effects associated with different types of macroplastic (>4.75 mm) was possible when linking entanglement rates to plastic debris exposure in the current work. As the upper boundary of the size category is left open, macroplastic includes a wide and heterogenous group of plastic with different properties associated with distinct fate and effect behaviors in the marine environment. Depending on the behavior and ecology of different species, their sensitivity to entanglement in plastic debris of different size and shape are variable (Woods et al., 2019). When this level of detail can be modelled, species life history traits such as body size which will influence what type of plastic that causes entanglement for that species can be integrated, and EC5's and subsequent SSD's can be made for distinct groups of macroplastic. For example, as fishing gear is frequently identified as the main entangling material for many species, the inclusion of a size category that encompass large plastic debris such as nets and other gear could capture the elevated risk associated with this specific product group. This also highlights another limitation with the current approach, which is that of distinguishing between entanglements caused my marine plastic debris and active fishing gear. This is often not feasible in entanglement studies, and if a large share of entanglements are indeed caused by active fishing gear rather than debris, there is an additional mismatch between the entanglement data and plastic debris concentration models that needs to be accounted for in future work.

Although this thesis contributes with data that can be used in species sensitivity distributions (SSDs), there is still a need for more standardized entanglement data to be included in dose-response modelling for extrapolating the EC5s. Not only are additional datapoints pertaining the species represented in this work needed, but also other species groups prone to entanglements, such as sea turtles (Duncan et al., 2017), in order to obtain a better representation of the marine ecosystem in the analyses. Finally, for the inclusion of the environmental impacts of mismanaged plastic waste in the LCA framework, effect factors for ingestion and other impact pathways of plastic debris in different size categories also needs to be developed.

6 References

ADIMEY, N. M., HUDAK, C. A., POWELL, J. R., BASSOS-HULL, K., FOLEY, A., FARMER, N. A., WHITE, L. & MINCH, K. 2014. Fishery gear interactions from stranded bottlenose dolphins, Florida manatees and sea turtles in Florida, U.S.A. *Marine Pollution Bulletin,* 81**,** 103-115.

ALLEN, R., JARVIS, D., SAYER, S. & MILLS, C. 2012. Entanglement of grey seals Halichoerus grypus at a haul out site in Cornwall, UK. *Marine Pollution Bulletin,* 64**,** 2815-2819.

ANDERWALD, P., DANÍELSDÓTTIR, A. K., HAUG, T., LARSEN, F., LESAGE, V., REID, R. J., VÍKINGSSON, G. A. & HOELZEL, A. R. 2011. Possible cryptic stock structure for minke whales in the North Atlantic: Implications for conservation and management. *Biological Conservation,* 144**,** 2479-2489.

ANDRADY, A. L. 2015. Persistence of Plastic Litter in the Oceans. *In:* BERGMANN, M., GUTOW, L. & KLAGES, M. (eds.) *Marine Anthropogenic Litter.* Cham: Springer International Publishing.

ARNOULD, J. P. Y. & KIRKWOOD, R. 2007. Habitat selection by female Australian fur seals (*Arctocephalus pusillus doriferus*). *Aquatic Conservation: Marine and Freshwater Ecosystems,* 17**,** S53-S67.

ARTHUR, B., HINDELL, M., BESTER, M., DE BRUYN, N., TRATHAN, P., GOEBEL, M. & LEA, M.-A. 2017. Winter habitat predictions of a key Southern Ocean predator, the Antarctic fur seal (Arctocephalus gazella). *Deep Sea Research Part II: Topical Studies in Oceanography,* 140**,** 171-181.

ARTHUR, B., HINDELL, M., BESTER, M., TRATHAN, P., JONSEN, I., STANILAND, I., OOSTHUIZEN, W. C., WEGE, M. & LEA, M.-A. 2015. Return Customers: Foraging Site Fidelity and the Effect of Environmental Variability in Wide-Ranging Antarctic Fur Seals. *PLOS ONE,* 10**,** e0120888.

ARTHUR, C., BAKER, J. E. & BAMFORD, H. A. 2009. Proceedings of the International Research Workshop on the Occurrence, Effects, and Fate of Microplastic Marine Debris *In:* ARTHUR, C., BAKER, J. E. & BAMFORD, H. A. (eds.). Tacoma, WA, USA: University of Washington Tacoma.

BALMER, B., ZOLMAN, E., ROWLES, T., SMITH, C., TOWNSEND, F., FAUQUIER, D., GEORGE, C., GOLDSTEIN, T., HANSEN, L., QUIGLEY, B., MCFEE, W., MOREY, J., ROSEL, P., SALIKI, J., SPEAKMAN, T. & SCHWACKE, L. 2018. Ranging patterns, spatial overlap, and association with dolphin morbillivirus exposure in common bottlenose dolphins (Tursiops truncatus) along the Georgia, USA coast. *Ecology and Evolution,* 8**,** 12890-12904.

BARNES, D. K. A., GALGANI, F., THOMPSON, R. C. & BARLAZ, M. 2009. Accumulation and fragmentation of plastic debris in global environments. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences,* 364**,** 1985-1998.

BAUMGARTNER, M. F., WENZEL, F. W., LYSIAK, N. S. J. & PATRICIAN, M. R. 2017. North Atlantic right whale foraging ecology and its role in human-caused mortality. *Marine Ecology Progress Series,* 581.

BIRDLIFE INTERNATIONAL 2015. *Fulmarus glacialis*. *The IUCN Red List of Threatened Species* 2015.

BIRDLIFE INTERNATIONAL 2018a. *Fulmarus glacialis*. *The IUCN Red List of Threatened Species,* 2018.

BIRDLIFE INTERNATIONAL 2018b. *Morus bassanus*. *The IUCN Red List of Threatened Species,* 2018.

BOND, A. L., MONTEVECCHI, W. A., GUSE, N., REGULAR, P. M., GARTHE, S. & RAIL, J.-F. 2012. Prevalence and composition of fishing gear debris in the nests of northern gannets (Morus bassanus) are related to fishing effort. *Marine Pollution Bulletin,* 64**,** 907-911.

BOND, T., FERRANDIZ-MAS, V., FELIPE-SOTELO, M. & VAN SEBILLE, E. 2018. The occurrence and degradation of aquatic plastic litter based on polymer physicochemical properties: A review. *Critical Reviews in Environmental Science and Technology,* 48**,** 685-722.

BONIN, C. A. 2012. *Population genetics and mating system of Antarctic fur seals, Arctocephalus gazella, at Livingston Island, Antarctica.* PhD, UC San Diego.

BOOR, G. K. H. & SMALL , R. J. 2012. Steller sea lion spatial-use patterns derived from a Bayesian model of opportunistic observations. *Marine Mammal Science,* 28**,** E375-E403.

BOOTH, A. M., KUBOWICZ, S., BEEGLE-KRAUSE, C., SKANCKE, J., NORDAM, T., LANDSEM, E., THRONE-HOLST, M. & JAHREN, S. 2017. Microplastic in global and Norwegian marine environments: Distributions, degradation mechanisms and transport. Trondheim, Norway: SINTEF.

BOUCHER, J., DUBOIS, C., KOUNINA, A. & PUYDARRIEUX, P. 2019. *Review of plastic footprint methodologies: Laying the foundation for the development of a standardised plastic footprint measurement tool,* Gland, Switzerland, IUCN.

BOWEN, D. 2016. Halichoerus grypus. *The IUCN Red List of Threatened Species,* 2016.

BOYD, I., MCCAFFERTY, D., REID, K., TAYLOR, R. & WALKER, T. 1998. Dispersal of male and female Antarctic seals (Arctocephalus gazella). *Canadian Journal of Fisheries and Aquatic Sciences,* 55**,** 845-852.

BREED, G. A., JONSEN, I., MYERS, R., BOWEN, W. & LEONARD, M. 2009. Sex-specific, seasonal foraging tactics of adult grey seals (Halichoerus grypus) revealed by state-space analysis. *Ecology,* 90**,** 3209-21.

BREEN, P., BROWN, S., REID, D. & ROGAN, E. 2016. Modelling cetacean distribution and mapping overlap with fisheries in the northeast Atlantic. *Ocean & Coastal Management,* 134**,** 140-149.

BRILLANT, S., VANDERLAAN, A., RANGELEY, R. & TAGGART, C. 2015. Quantitative estimates of the movement and distribution of North Atlantic right whales along the northeast coast of North America. *Endangered Species Research,* 27**,** 141-154.

BURG, T., LOMAX, J., ALMOND, R., BROOKE, M. & AMOS, W. 2003. Unravelling dispersal patterns in an expanding population of a highly mobile seabird, the northern fulmar (Fulmarus glacialis). *Proceedings. Biological sciences / The Royal Society,* 270**,** 979-84.

CAMPHUYSEN, C. J. & VAN DER MEER, J. 2005. Wintering seabirds in West Africa: foraging hotspots off Western Sahara and Mauritania driven by upwelling and fisheries. *African Journal of Marine Science,* 27**,** 427-437.

CAMPHUYSEN, K. & GARTHE, S. 1997. An evaluation of the distribution and scavenging habits of northern fulmars (Fulmarus glacialis) in the North Sea. *ICES Journal of Marine Science,* 54**,** 654-683.

CARTER, M. I. D., RUSSELL, D. J. F., EMBLING, C. B., BLIGHT, C. J., THOMPSON, D., HOSEGOOD, P. J. & BENNETT, K. A. 2017. Intrinsic and extrinsic factors drive ontogeny of early-life at-sea behaviour in a marine top predator. *Scientific Reports,* 7**,** 15505.

CASTELBLANCO-MARTÍNEZ, D. N., BERMÚDEZ-ROMERO, A. L., GÓMEZ-CAMELO, I. V., ROSAS, F. C. W., TRUJILLO, F. & ZERDA-ORDOÑEZ, E. 2009. Seasonality of habitat use, mortality and reproduction of the vulnerable Antillean manatee *Trichechus manatus manatusin* the Orinoco River, Colombia: Implications for conservation. . *Oryx,* 43**,** 235-242.

COE, J. M. & ROGERS, D. B. 1997. *Marine Debris: Sources, Impacts, and Solutions*, Springer.

COLE, T., HAMILTON, P., HENRY, A., DULEY, P., PACE, R., WHITE, B. & FRASIER, T. 2013. Evidence of a North Atlantic right whale Eubalaena glacialis mating ground. *Endangered Species Research,* 21.

COMPA, M., ALOMAR, C., WILCOX, C., VAN SEBILLE, E., LEBRETON, L., HARDESTY, B. D. & DEUDERO, S. 2019. Risk assessment of plastic pollution on marine diversity in the Mediterranean Sea. *Science of The Total Environment,* 678**,** 188-196.

CONRADT, S. 2016. Der Basstölpel - Seevogel des Jahres 2016 - Der Fluch des billigen Plastiks. *Seevögel,* 37**,** 4-13.

COOKE, J. G. 2018. *Balaenoptera acutorostrata*. *The IUCN Red List of Threatened Species,* 2018.

CORDES, L. S., HEDWORTH, H. E., CABOT, D., CASSIDY, M. & THOMPSON, P. M. 2015. Parallel declines in survival of adult Northern Fulmars Fulmarus glacialis at colonies in Scotland and Ireland. *Ibis,* 157**,** 631-636.

CORNICK, L., NEILL, W. & GRANT, W. 2006. Assessing competition between Steller sea lions and the commercial groundfishery in Alaska: A bioenergetics modelling approach. *Ecological Modelling,* 199**,** 107-114.

CRONIN, M., GERRITSEN, H., REID, D. & JESSOPP, M. 2016. Spatial Overlap of Grey Seals and Fisheries in Irish Waters, Some New Insights Using Telemetry Technology and VMS. *PloS one,* 11**,** e0160564-e0160564.

CRONIN, M., POMEROY, P. & JESSOPP, M. 2012. Size and seasonal influences on the foraging range of female grey seals in the northeast Atlantic. *Marine Biology,* 160.

CURRAN, M. A. 2008. Life-Cycle Assessment. *In:* JØRGENSEN, S. E. & FATH, B. D. (eds.) *Encyclopedia of Ecology.* Oxford: Academic Press.

DEAVILLE, R., P.D., J., BROWNLOW, A., REID, R. J., SMITH, B., DUFFELL, E. L., C., S. R., PENROSE, R. & PERKINS, M. 2011. Final report for the period 1st January 2005 - 31st December 2010 *UK Cetacean Strandings Investigation Programme.*

DEAVILLE, R., P.D., J., PERKINS, M., BROWNLOW, A., DAVISON, N., TEN DOESCHATE, M., SMITH, B., LYAL, R., ALLAN, L., C., S. R. & PENROSE, R. 2016. Annual report for the period 1st January - 31st December 2015 *UK Cetacean Strandings Investigation Programme.* MB0111 Marine Biodiversity Division, Defra.

DEFRAN, R. H., WELLER, D. W., KELLY, D. L. & ESPINOSA, M. A. 1999. RANGE CHARACTERISTICS OF PACIFIC COAST BOTTLENOSE DOLPHINS (TURSIOPS TRUNCATUS) IN THE SOUTHERN CALIFORNIA BIGHT. *Marine Mammal Science,* 15**,** 381-393.

DEUTSCH, C. J., SELF-SULLIVAN, C. & MIGNUCCI-GIANNONI, A. 2008. *Trichechus manatus*. *The IUCN Red List of Threatened Species,* 2008.

DIERSCHKE, V., GARTHE, S. & MARKONES, N. 2004. Aktionsradien Helgoländer Dreizehenmöwen Rissa tridactyla und Trottellummen Uria aalge während der Aufzuchtphase. *Vogelwelt,* 125**,** 11-19.

DOLMAN, S. J. & MOORE, M. J. 2017. Welfare Implications of Cetacean Bycatch and Entanglements. *In:* BUTTERWORTH, A. (ed.) *Marine Mammal Welfare: Human Induced Change in the Marine Environment and its Impacts on Marine Mammal Welfare.* Cham: Springer International Publishing.

DUNCAN, E., BOTTERELL, Z., BRODERICK, A. C., GALLOWAY, T., LINDEQUE, P., NUNO, A. & GODLEY, B. 2017. A global review of marine turtle entanglement in anthropogenic debris: A baseline for further action. *Endangered Species Research,* 34.

EDWARDS, E. W. J., QUINN, L. R. & THOMPSON, P. M. 2016a. State-space modelling of geolocation data reveals sex differences in the use of management areas by breeding northern fulmars. *Journal of Applied Ecology,* 53**,** 1880-1889.

EDWARDS, E. W. J., QUINN, L. R., WAKEFIELD, E. D., MILLER, P. I. & THOMPSON, P. M. 2013. Tracking a northern fulmar from a Scottish nesting site to the Charlie-Gibbs Fracture Zone: Evidence of linkage between coastal breeding seabirds and Mid-Atlantic Ridge feeding sites. *Deep Sea Research Part II: Topical Studies in Oceanography,* 98**,** 438-444.

EDWARDS, H. H., MARTIN, J., DEUTSCH, C. J., MULLER, R. G., KOSLOVSKY, S. M., SMITH, A. J. & BARLAS, M. E. 2016b. Influence of Manatees' Diving on Their Risk of Collision with Watercraft. *PloS one,* 11**,** e0151450-e0151450.

ELLIOTT, K. H. & GASTON, A. J. 2005. Flight speeds of two seabirds: a test of Norberg's hypothesis. *Ibis,* 147**,** 783-789.

ERIKSEN, M., LEBRETON, L. C. M., CARSON, H. S., THIEL, M., MOORE, C. J., BORERRO, J. C., GALGANI, F., RYAN, P. G. & REISSER, J. 2014. Plastic Pollution in the World's Oceans: More than 5 Trillion Plastic Pieces Weighing over 250,000 Tons Afloat at Sea. *PLOS ONE,* 9**,** e111913.

FAO 2016. Abandoned, lost or otherwise discarded gillnets and trammel nets: methods to estimate ghost fishing mortality, and the status of regional monitoring and management. *In:* GILMAN, E., CHOPIN, F., SUURONEN, P. & KUEMLANGAN, B. (eds.) *FAO Fisheries and Aquaculture Technical Paper No. 600.* Rome, Italy.

FAZEY, F. M. C. & RYAN, P. G. 2016. Debris size and buoyancy influence the dispersal distance of stranded litter. *Marine Pollution Bulletin,* 110**,** 371-377.

FIFIELD, D., MONTEVECCHI, W., GARTHE, S., ROBERTSON, G., KUBETZKI, U. & RAIL, J.-F. 2014. Migratory tactics and wintering areas of Northern Gannets (Morus bassanus) breeding in North America. *Ornithological Monographs,* 79**,** 1-63.

FILELLA, M. 2015. Questions of size and numbers in environmental research on microplastics: methodological and conceptual aspects. *Environmental Chemistry,* 12**,** 527–538.

FIRESTONE, J., LYONS, S., WANG, C. & CORBETT, J. 2008. Statistical modeling of North Atlantic right whale migration along the mid-Atlantic region of the eastern seaboard of the United States. *Biological Conservation,* 141**,** 221-232.

FLAMM, R. O., WEIGLE, B. L., WRIGHT, I. E., ROSS, M. & AGLIETTI, S. 2005. ESTIMATION OF MANATEE (TRICHECHUS MANATUS LATIROSTRIS) PLACES AND MOVEMENT CORRIDORS USING TELEMETRY DATA. *Ecological Applications,* 15**,** 1415-1426.

FORT, J., PETTEX, E., TREMBLAY, Y., LORENTSEN, S.-H., GARTHE, S., VOTIER, S., PONS, J. B., SIORAT, F., FURNESS, R. W., GRECIAN, W. J., BEARHOP, S., MONTEVECCHI, W. A. & GRÉMILLET, D. 2012. Meta-population evidence of oriented chain migration in northern gannets (Morus bassanus). *Frontiers in Ecology and the Environment,* 10**,** 237-242.

GALGANI, F., HANKE, G., WERNER, S., L, O., NILSSON, P., FLEET, D., KINSEY, S., RC, T., VAN FRANEKER, J., VLACHOGIANNI, T., SCOULLOS, M., MIRA VEIGA, J., PALATINUS, A., MATIDDI, M., MAES, T., KORPINEN, S., A, B., H, L., GAGO, J. & G, L. 2013. *Guidance on Monitoring of Marine Litter in European Seas*.

GALL, S. C. & THOMPSON, R. C. 2015. The impact of debris on marine life. *Marine Pollution Bulletin,* 92**,** 170-179.

GARRISON, L. P., BARRY, K. & W. HOGGARD, W. 2017. The abundance of coastal morphotype bottlenose dolphins along the U.S. east coast: 2002-2016. Miami, Florida: Southeast Fisheries Science Center.

GARTHE, S. 2019. *Counting seabirds from ships and aircraft* [Online]. Available: http://www.coastalwiki.org/wiki/Counting_seabirds_from_ships_and_aircraft [Accessed 01.05 2020].

GARTHE, S., HALLGRIMSSON, G. T., MONTEVECCHI, W. A., FIFIELD, D. & FURNESS, R. W. 2016. East or west? Migration routes and wintering sites of Northern Gannets Morus bassanus from south-eastern Iceland. *Marine Biology,* 163**,** 151.

GARTHE, S., LUDYNIA, K., HÜPPOP, O., KUBETZKI, U., MERAZ, J. F. & FURNESS, R. W. 2012. Energy budgets reveal equal benefits of varied migration strategies in northern gannets. *Marine Biology,* 159**,** 1907-1915.

GARTHE, S., MARKONES, N. & CORMAN, A.-M. 2017a. Possible impacts of offshore wind farms on seabirds: a pilot study in Northern Gannets in the southern North Sea. *Journal of Ornithology,* 158**,** 345-349.

GARTHE, S., MARKONES, N., HÜPPOP, O. & ADLER, S. 2009. Effects of hydrographic and meteorological factors on seasonal seabird abundance in the southern North Sea. *Marine Ecology-progress Series - MAR ECOL-PROGR SER,* 395.

GARTHE, S., PESCHKO, V., KUBETZKI, U. & CORMAN, A.-M. 2017b. Seabirds as samplers of the marine environment - A case study of northern gannets. *Ocean Science,* 13**,** 337-347.

GASTON, A. J. 2004. *Seabirds: A Natural History,* New Haven, Connecticutt USA, Yale University Press.

GEBCO 2020. General Bathymetric Chart of the Oceans Compilation Group 2020 Grid.

GELATT, T. & SWEENEY, K. 2016a. *Eumetopias jubatus*. *The IUCN Red List of Threatened Species,* 2016.

GELATT, T. & SWEENEY, K. 2016b. *Eumetopias jubatus* ssp. *monteriensis*. *The IUCN Red List of Threatened Species* 2016.

GESAMP 2015. Sources, fate and effects of microplastics in the marine environment: a global assessment. *In:* KERSHAW, P. (ed.) *IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection.* Rep. Stud. GESAMP.

GESAMP 2019. Guidelines for the monitoring & assessment of plastic litter in the ocean. *In:* PETER KERSHAW, A. T., FRANCOIS GALGANI (ed.) *IMO/FAO/UNESCO-IOC/UNIDO/WMO/IAEA/UN/UNEP/UNDP/ISA Joint Group of Experts on the Scientific Aspects of Marine Environmental Protection* Rep. Stud. GESAMP

GEYER, R., JAMBECK, J. R. & LAW, K. L. 2017. Production, use, and fate of all plastics ever made. *Science Advances,* 3**,** e1700782.

GILARDI, K. V. K., CARLSON-BREMER, D., JUNE, J. A., ANTONELIS, K., BROADHURST, G. & COWAN, T. 2010. Marine species mortality in derelict fishing nets in Puget Sound, WA and the cost/benefits of derelict net removal. *Marine Pollution Bulletin,* 60**,** 376-382.

GOOD, C. P. 2008. *Spatial ecology of the north atlantic right whale (eubalaena glacialis).* PhD, Duke University.

GOSCH, M., CRONIN, M., ROGAN, E., HUNT, W., LUCK, C. & JESSOPP, M. 2019. Spatial variation in a top marine predator’s diet at two regionally distinct sites. *PLOS ONE,* 14**,** e0209032.

GOWAN, T. A. & ORTEGA-ORTIZ, J. G. 2014. Wintering Habitat Model for the North Atlantic Right Whale (Eubalaena glacialis) in the Southeastern United States. *PLOS ONE,* 9**,** e95126.

GREGORY, M. R. 2009. Environmental implications of plastic debris in marine settings--entanglement, ingestion, smothering, hangers-on, hitch-hiking and alien invasions. *Philosophical transactions of the Royal Society of London. Series B, Biological sciences,* 364**,** 2013-2025.

GRÉMILLET, D., PÉRON, C., PROVOST, P. & LESCROEL, A. 2015. Adult and juvenile European seabirds at risk from marine plundering off West Africa. *Biological Conservation,* 182.

HAMER, K. C., PHILLIPS, R. A., WANLESS, S., HARRIS, M. P. & WOOD, A. G. 2000. Foraging ranges, diets and feeding locations of gannets Morus bassanus in the North Sea: evidence from satellite telemetry. *Marine Ecology Progress Series,* 200**,** 257-264.

HAMER, K. C., THOMPSON, D. R. & GRAY, C. M. 1997. Spatial variation in the feeding ecology, foraging ranges, and breeding energetics of northern fulmars in the north-east Atlantic Ocean. *ICES Journal of Marine Science,* 54**,** 645-653.

HAMMOND, P., NORTHRIDGE, S., THOMPSON, D., GORDON, J. C. D., HALL, A. J., DUCK, C. D., AARTS, G., CUNNINGHAM, L., EMBLING, C. B. & MATTHIOPOULOS, J. 2006. Background information on marine mammals relevant to Strategic Environmental Assessment Area 7 (SEA 7). *Technical report to the Department of Trade and Industry.* St Andrews, Fife KY16 8LB, Scotland: Sea Mammal Research Unit, University of St Andrews.

HARCOURT, R., VAN DER HOOP, J., KRAUS, S. & CARROLL, E. L. 2019. Future Directions in Eubalaena spp.: Comparative Research to Inform Conservation. *Frontiers in Marine Science,* 5.

HARTMANN, N. B., HÜFFER, T., THOMPSON, R. C., HASSELLÖV, M., VERSCHOOR, A., DAUGAARD, A. E., RIST, S., KARLSSON, T., BRENNHOLT, N., COLE, M., HERRLING, M. P., HESS, M. C., IVLEVA, N. P., LUSHER, A. L. & WAGNER, M. 2019. Are We Speaking the Same Language? Recommendations for a Definition and Categorization Framework for Plastic Debris. *Environmental Science & Technology,* 53**,** 1039-1047.

HASTINGS, K. K., REHBERG, M. J., O'CORRY-CROWE, G. M., PENDLETON, G. W., JEMISON, L. A. & GELATT, T. S. 2019. Demographic consequences and characteristics of recent population mixing and colonization in Steller sea lions, Eumetopias jubatus. *Journal of Mammalogy,* 101**,** 107-120.

HAUSCHILD, M. & HUIJBREGTS, M. 2015. Introducing Life Cycle Impact Assessment.

HERZING, D. & ELLISER, C. R. 2016. Opportunistic Sightings of Cetaceans in Nearshore and Offshore Waters of Southeast Florida. *Journal of Northwest Atlantic Fishery Science,* 48**,** 21-31.

HIDALGO-RUZ, V., GUTOW, L., THOMPSON, R. & THIEL, M. 2012. Microplastics in the Marine Environment: A Review of the Methods Used for Identification and Quantification. *Environmental science & technology,* 46**,** 3060-75.

HOFMEYR, G. J. G. 2015. *Arctocephalus pusillus*. *The IUCN Red List of Threatened Species,* 2015.

HOFMEYR, G. J. G. 2016. *Arctocephalus gazella*. *The IUCN Red List of Threatened Species,* 2016.

HOFMEYR, G. J. G., BESTER, M. N., KIRKMAN, S. P., LYDERSEN, C. & KOVACS, K. M. 2006. Entanglement of Antarctic fur seals at Bouvetøya, Southern Ocean. *Marine Pollution Bulletin,* 52**,** 1077-1080.

HOSKINS, A. J., SCHUMANN, N., COSTA, D. P. & ARNOULD, J. P. Y. 2017. Foraging niche separation in sympatric temperate-latitude fur seal species. *Marine Ecology Progress Series,* 566**,** 229-241.

HUMBERT, S., ROSSI, V., MARGNI, M., JOLLIET, O. & LOERINCIK, Y. 2009. Life cycle assessment of two baby food packaging alternatives: glass jars vs. plastic pots. *The International Journal of Life Cycle Assessment,* 14**,** 95-106.

HUNT, K., ROLLAND, R. & KRAUS, S. 2015. Conservation Physiology of an Uncatchable Animal: The North Atlantic Right Whale (Eubalaena glacialis). *Integrative and comparative biology,* 55.

IUCN, T. I. U. F. C. O. N. 2020. *The IUCN Red List of Threatened Species: Spatial Data Download: http://www.iucnredlist.org/technicaldocuments/spatial-data* [Online]. [Accessed].

JACOBSEN, K.-O., MARX, M. & ØIEN, N. 2004. TWO-WAY TRANS-ATLANTIC MIGRATION OF A NORTH ATLANTIC RIGHT WHALE (EUBALAENA GLACIALIS). *Marine Mammal Science,* 20**,** 161-166.

JAMBECK, J. R., GEYER, R., WILCOX, C., SIEGLER, T. R., PERRYMAN, M., ANDRADY, A., NARAYAN, R. & LAW, K. L. 2015. Plastic waste inputs from land into the ocean. *Science,* 347**,** 768-771.

JEMISON, L. A., PENDLETON, G. W., FRITZ, L. W., HASTINGS, K. K., MANISCALCO, J. M., TRITES, A. W. & GELATT, T. S. 2013. Inter-Population Movements of Steller Sea Lions in Alaska with Implications for Population Separation. *PLOS ONE,* 8**,** e70167.

JEMISON, L. A., PENDLETON, G. W., HASTINGS, K. K., MANISCALCO, J. M. & FRITZ, L. W. 2018. Spatial distribution, movements, and geographic range of Steller sea lions (Eumetopias jubatus) in Alaska. *PLOS ONE,* 13**,** e0208093.

JOLLIET, O., ANTÓN, A., BOULAY, A.-M., CHERUBINI, F., FANTKE, P., LEVASSEUR, A., MCKONE, T. E., MICHELSEN, O., CANALS, L. M. I., MOTOSHITA, M., PFISTER, S., VERONES, F., VIGON, B. & FRISCHKNECHT, R. 2018. Global guidance on environmental life cycle impact assessment indicators: impacts of climate change, fine particulate matter formation, water consumption and land use *The International Journal of Life Cycle Assessment,* 23**,** 2189-2207.

KIM, S.-G., LEE, W.-I. L. & YUSEOK, M. 2014. The estimation of derelict fishing gear in the coastal waters of South Korea: Trap and gill-net fisheries. *Marine Policy,* 46**,** 119-122.

KIRKWOOD, R. & ARNOULD, J. 2011. Foraging trip strategies and habitat use during late pup rearing by lactating Australian fur seals. *Australian Journal of Zoology,* 59**,** 216-226.

KIRKWOOD, R. & GOLDSWORTHY, S. 2013. *Fur Seals and Sea Lions*.

KIRKWOOD, R., LYNCH, M., GALES, N., DANN, P. & SUMNER, M. 2007. At-sea movements and habitat use of adult male Australian fur seals (Arctocephalus pusillus doriferus). *Canadian Journal of Zoology,* 84**,** 1781-1788.

KIRKWOOD, R., PEMBERTON, D., GALES, R., HOSKINS, A., MITCHELL, T., SHAUGHNESSY, P. & ARNOULD, J. 2010. Continued population recovery by Australian fur seals. *Marine and Freshwater Research,* 61**,** 695-701.

KNOWLTON, A. R., SIGUKJÓSSON, J., CIANO, J. N. & KRAUS, S. D. 1992. LONG-DISTANCE MOVEMENTS OF NORTH ATLANTIC RIGHT WHALES (EUBALAENA GLACIALIS). *Marine Mammal Science,* 8**,** 397-405.

KNOX, T., BAYLIS, A. & ARNOULD, J. 2017. Habitat use and diving behaviour of male Australian fur seals. *Marine Ecology Progress Series,* 566.

KNOX, T. C., BAYLIS, A. M. M. & ARNOULD, J. P. Y. 2018. Foraging site fidelity in male Australian fur seals. *Marine Biology,* 165.

KOT, B. W., SEARS, R., ANIS, A., NOWACEK, D. P., GEDAMKE, J. & MARSHALL, C. D. 2012. Behavioral responses of minke whales (Balaenoptera acutorostrata) to experimental fishing gear in a coastal environment. *Journal of Experimental Marine Biology and Ecology,* 413**,** 13-20.

KRAUS, S. D., KENNEY, R. D., MAYO, C. A., MCLELLAN, W. A., MOORE, M. J. & NOWACEK, D. P. 2016. Recent Scientific Publications Cast Doubt on North Atlantic Right Whale Future. *Frontiers in Marine Science,* 3.

KUBETZKI, U., GARTHE, S., FIFIELD, D., MENDEL, B. & FURNESS, R. 2009. Individual migratory schedules and wintering areas of northern gannets. *Marine Ecology Progress Series,* 391**,** 257-265.

KÜHN, S., BRAVO REBOLLEDO, E. L. & VAN FRANEKER, J. A. 2015. Deleterious Effects of Litter on Marine Life. *In:* BERGMANN, M., GUTOW, L. & KLAGES, M. (eds.) *Marine Anthropogenic Litter.* Cham: Springer International Publishing.

LAIST, D. & REYNOLDS, J. E. 2005. Florida Manatees, Warm-Water Refuges, and an Uncertain Future. *Coastal Management,* 33**,** 279-295.

LAIST, D. W. 1987. Overview of the biological effects of lost and discarded plastic debris in the marine environment. *Marine Pollution Bulletin,* 18**,** 319-326.

LAIST, D. W. 1997. Impacts of Marine Debris: Entanglement of Marine Life in Marine Debris Including a Comprehensive List of Species with Entanglement and Ingestion Records. *In:* COE, J. M. & ROGERS, D. B. (eds.) *Marine Debris: Sources, Impacts, and Solutions.* New York, NY: Springer New York.

LANGTIMM, C., DORAZIO, R., STITH, B. & DOYLE, T. 2011. New Aerial Survey and Hierarchical Model to Estimate Manatee Abundance. *The Journal of Wildlife Management,* 75**,** 399-412.

LAW, K. L. 2017. Plastics in the Marine Environment. *Annual Review of Marine Science,* 9**,** 205-229.

LAWSON, T. J., WILCOX, C., JOHNS, K., DANN, P. & HARDESTY, B. D. 2015. Characteristics of marine debris that entangle Australian fur seals (Arctocephalus pusillus doriferus) in southern Australia. *Marine Pollution Bulletin,* 98**,** 354-357.

LEBRETON, L. & ANDRADY, A. 2019. Future scenarios of global plastic waste generation and disposal. *Palgrave Communications,* 5**,** 6.

LEBRETON, L., SLAT, B., FERRARI, F., SAINTE-ROSE, B., AITKEN, J., MARTHOUSE, R., HAJBANE, S., CUNSOLO, S., SCHWARZ, A., LEVIVIER, A., NOBLE, K., DEBELJAK, P., MARAL, H., SCHOENEICH-ARGENT, R., BRAMBINI, R. & REISSER, J. 2018. Evidence that the Great Pacific Garbage Patch is rapidly accumulating plastic. *Scientific Reports,* 8**,** 4666.

LEENEY, R. H., BRODERICK, A. C., MILLS, C., SAYER, S., WITT, M. J. & GODLEY, B. J. 2010. Abundance, distribution and haul-out behaviour of grey seals (Halichoerus grypus) in Cornwall and the Isles of Scilly, UK. *Journal of the Marine Biological Association of the United Kingdom,* 90**,** 1033-1040.

LEFEBVRE, L. W., REID, J. P., KENWORTHY, W. J. & POWELL, J. A. 2000. Characterizing manatee habitat use and seagrass grazing in Florida and Puerto Rico: Implications for conservation and management. *Pacific Conservation Biology,* 5**,** 289-298.

LEOPOLD, M., GRUNSKY, B., HÜPPOP, O., MAUL, A. & VAN DER MEER, J. 1995. How large an area of sea do Helgoland seabirds use for foraging during the breeding season? *Helgoländer Meeresuntersuchungen,* 49.

LEOPOLD, M., WOLF, P. & HÜPPOP, O. 1992. Food of young and colony-attendance of adult guillemots Uria aalge on Helgoland. *Helgoländer Meeresuntersuchungen,* 46**,** 237-249.

LOUGHLIN, T. 1997. Using the phylogeographic method to identify Steller sea lion stocks. *Molecular Genetics of Marine Mammals***,** 159-171.

LOUGHLIN, T. R. 2009. Steller Sea Lion: Eumetopias jubatus. *In:* PERRIN, W. F., WÜRSIG, B. & THEWISSEN, J. G. M. (eds.) *Encyclopedia of Marine Mammals (Second Edition).* London: Academic Press.

LUSHER, A. L., HERNANDEZ-MILIAN, G., BERROW, S., ROGAN, E. & O'CONNOR, I. 2018. Incidence of marine debris in cetaceans stranded and bycaught in Ireland: Recent findings and a review of historical knowledge. *Environ Pollut,* 232**,** 467-476.

LYSIAK, N. S. J., TRUMBLE, S. J., KNOWLTON, A. R. & MOORE, M. J. 2018. Characterizing the Duration and Severity of Fishing Gear Entanglement on a North Atlantic Right Whale (Eubalaena glacialis) Using Stable Isotopes, Steroid and Thyroid Hormones in Baleen. *Frontiers in Marine Science,* 5.

MACLEOD, K., FAIRBAIRNS, R., GILL, A., FAIRBAIRNS, B., GORDON, J., BLAIR-MYERS, C. & PARSONS, E. 2004. Seasonal distribution of minke whales Balaenoptera acutorostrata in relation to physiography and prey off the Isle of Mull, Scotland. *Marine Ecology Progress Series,* 277**,** 263-274.

MALLORY, M. L., HATCH, S. A. & NETTLESHIP, D. N. 2012. Northern Fulmar *(Fulmarus glacialis*), version 2.0. *In:* POOLE, A. F. (ed.) *The Birds of North America.* Ithaca, NY, USA: Cornell Lab of Ornithology.

MARKONES, N., DIERSCHKE, V. & GARTHE, S. 2010. Seasonal differences in at-sea activity of seabirds underline high energetic demands during the breeding period. *Journal of Ornithology,* 151**,** 329-336.

MARTIN, J., EDWARDS, H. H., FONNESBECK, C. J., KOSLOVSKY, S. M., HARMAK, C. W. & DANE, T. M. 2015. Combining information for monitoring at large spatial scales: First statewide abundance estimate of the Florida manatee. *Biological Conservation,* 186**,** 44-51.

MARTINS, M. C. I., SETTE, L., JOSEPHSON, E., BOGOMOLNI, A., ROSE, K., SHARP, S. M., NIEMEYER, M. & MOORE, M. 2019. Unoccupied aerial system assessment of entanglement in Northwest Atlantic gray seals (Halichoerus grypus). *Marine Mammal Science,* 35**,** 1613-1624.

MATE, B. R., NIEUKIRK, S. L. & KRAUS, S. D. 1997. Satellite-Monitored Movements of the Northern Right Whale. *The Journal of Wildlife Management,* 61**,** 1393-1405.

MATSUOKA, T., NAKASHIMA, T. & NAGASAWA, N. 2005. A review of ghost fishing: scientific approaches to evaluation and solutions. *Fisheries Science,* 71**,** 691-702.

MCCONNELL, B. J., FEDAK, M. A., LOVELL, P. & HAMMOND, P. S. 1999. Movements and foraging areas of grey seals in the North Sea. *Journal of Applied Ecology,* 36**,** 573-590.

MCHARDY, C. 2019. *Linking marine plastic debris quantities to entanglement rate: development of a life cycle impact assessment effect factor based on species sensitivity.* Master thesis, Norwegian University of Science and Technology.

MCINTOSH, R. R., KIRKWOOD, R., SUTHERLAND, D. R. & DANN, P. 2015. Drivers and annual estimates of marine wildlife entanglement rates: A long-term case study with Australian fur seals. *Marine Pollution Bulletin,* 101**,** 716-725.

MENDEL, B., KOTZERKA, J., SOMMERFELD, J., SCHWEMMER, H., SONNTAG, N. & GARTHE, S. 2014. Effects of the alpha ventus offshore test site on distribution patterns, behaviour and flight heights of seabirds. *In:* FEDERAL, M., HYDROGRAPHIC, A., FEDERAL MINISTRY FOR THE ENVIRONMENT, N. C. & NUCLEAR, S. (eds.) *Ecological Research at the Offshore Windfarm alpha ventus: Challenges, Results and Perspectives.* Wiesbaden: Springer Fachmedien Wiesbaden.

MENDENHALL, E. 2018. Oceans of plastic: A research agenda to propel policy development. *Marine Policy,* 96.

MONAGHAN, P., WALTON, P., WANLESS, S., UTTLEY, J. D. & BLJRNS, M. D. 1994. Effects of prey abundance on the foraging behaviour, diving efficiency and time allocation of breeding Guillemots Uria aalge. *Ibis,* 136**,** 214-222.

MONTEVECCHI, W. 1991. Incidence and types of plastic in gannets' nests in the northwest Atlantic. *Canadian Journal of Zoology,* 69**,** 295-297.

NELMS, S. E., COOMBES, C., FOSTER, L. C., GALLOWAY, T. S., GODLEY, B. J., LINDEQUE, P. K. & WITT, M. J. 2017. Marine anthropogenic litter on British beaches: A 10-year nationwide assessment using citizen science data. *Science of The Total Environment,* 579**,** 1399-1409.

NOAA. 2016. *North Atlantic Right Whale Critical Habitat*. Gloucester, MA: NOAA National Marine Fisheries Service (NMFS) - Greater Atlantic Regional Fisheries Office (GARFO).

NOAA. 2020. *Marine Mammal Stock Assessment Reports by Species/Stock* [Online]. U. S. National Oceanic and Atmospheric Administration (NOAA) Fisheries. Available: https://www.fisheries.noaa.gov/national/marine-mammal-protection/marine-mammal-stock-assessment-reports-species-stock [Accessed 01.05.2020].

NORTHRIDGE, S. P., CARGILL, A., CORAM, A., MANDELBERG, L., S., C. & REID, B. 2010. Entanglement of minke whales in Scottish waters; aninvestigation into occurrence, causes and mitigation. *Report to Scottish Government.* St Andrews, Scotland: Sea Mammal Research Unit, University of St Andrews.

NORTHRIDGE, S. P., TASKER, M. L., WEBB, A. & WILLIAMS, J. M. 1995. Distribution and relative abundance of harbour porpoises (Phocoena phocoena L.), white-beaked dolphins (Lagenorhynchus albirostris Gray), and minke whales (Balaenoptera acutorostrata Lacepède) around the British Isles. *ICES Journal of Marine Science,* 52**,** 55-66.

O'HANLON, N., BOND, A., LAVERS, J., MASDEN, E. & JAMES, N. 2019. Monitoring nest incorporation of anthropogenic debris by Northern Gannets across their range. *Environmental Pollution,* 255**,** 113152.

OBIS. 2020. *Ocean Biogeographic Information System* [Online]. Available: www.obis.org [Accessed 01.05.2020].

OJOWSKI, U., EIDTMANN, C., FURNESS, R. W. & GARTHE, S. 2001. Diet and nest attendance of incubating and chick-rearing northern fulmars (Fulmarus glacialis) in Shetland. *Marine Biology,* 139**,** 1193-1200.

OKSANEN, S., AHOLA, M., LEHTONEN, E. & KUNNASRANTA, M. 2014. Using movement data of Baltic grey seals to examine foraging-site fidelity: Implications for seal-fishery conflict mitigation. *Marine Ecology Progress Series,* 507**,** 297-308.

OLIVERA-GÓMEZ, L. D. & MELLINK, E. 2005. Distribution of the Antillean manatee (*Trichechus manatus manatus*) as a function of habitat characteristics, in Bah́ia de Chetumal, Mexico. *Biological Conservation,,* 121**,** 127-133.

ORIANS, G. H. & PEARSON, N. E. 1977. *On the theory of central place foraging,* Columbus, Ohio State University Press.

PANTI, C., BAINI, M., LUSHER, A., HERNANDEZ-MILAN, G., BRAVO REBOLLEDO, E. L., UNGER, B., SYBERG, K., SIMMONDS, M. P. & FOSSI, M. C. 2019. Marine litter: One of the major threats for marine mammals. Outcomes from the European Cetacean Society workshop. *Environmental Pollution,* 247**,** 72-79.

PEMBERTON, D., BROTHERS, N. & KIRKWOOD, R. 1992. Entanglement of Australian fur seals in man-made debris in Tasmanian waters. *Wildlife Research,* 19**,** 151-159.

PERRIN, W. F., THIELEKING, J. L., WALKER, W. A., ARCHER, F. I. & ROBERTSON, K. M. 2011. Common bottlenose dolphins (Tursiops truncatus) in California waters: Cranial differentiation of coastal and offshore ecotypes. *Marine Mammal Science,* 27**,** 769-792.

PETTEX, E., LORENTSEN, S.-H., GRÉMILLET, D., GIMENEZ, O., BARRETT, R. T., PONS, J.-B., BOHEC, C. & BONADONNA, F. 2012. Multi-scale foraging variability in Northern gannet (Morus bassanus) fuels potential foraging plasticity. *Marine Biology*.

PETTIS, H. M., PACE, R. M. & HAMILTON, P. K. 2020. North Atlantic Right Whale Consortium 2019 Annual Report Card. *Report to the North Atlantic Right Whale Consortium.*

PITCHER, K. W., OLESIUK, P. F., BROWN, R. F., LOWRY, M. S., JEFFRIES, S. J., SEASE, J. L., PERRYMAN, W. L., STINCHCOMB, C. E. & LOWRY, L. F. 2007. Abundance and distribution of the eastern North Pacific Steller sea lion (Eumetopias jubatus) population. *Fishery Bulletin,* 105**,** 102+.

PLASTICSEUROPE. 2019. *Plastics – the Facts 2019 An analysis of European plastics production, demand and waste data* [Online]. Available: https://www.plasticseurope.org/en/resources/publications/1804-plastics-facts-2019 [Accessed 30.10 2019].

POMEROY, P. P., ANDERSON, S. S., TWISS, S. D. & MCCONNELL, B. J. 1994. Dispersion and site fidelity of breeding female grey seals (Halichoerus grypus) on North Rona, Scotland. *Journal of Zoology,* 233**,** 429-447.

RAUM-SURYAN, K. L., JEMISON, L. A. & PITCHER, K. W. 2009. Entanglement of Steller sea lions (*Eumetopias jubatus*) in marine debris: Identifying causes and finding solutions. *Marine Pollution Bulletin,* 58**,** 1487-1495.

RAUM-SURYAN, K. L., PITCHER, K. W., CALKINS, D. G., SEASE, J. L. & LOUGHLIN, T. R. 2002. Dispersal, rookery fidelity, and metapopulation structure of steller sea lions (*eumetopias jubatus*) in an increasing and a decreasing population in Alaska. *Marine Mammal Science,* 18**,** 746-764.

RAUM-SURYAN, K. L., REHBERG, M. J., PENDLETON, G. W., PITCHER, K. W. & GELATT, T. S. 2004. Development of dispersal, movement patterns, and haul-out use by pup and juvenile steller sea lions (*eumetopias jubatus*) in Alaska. *Marine Mammal Science,* 20**,** 823-850.

RECIPE 2009. A life cycle impact assessment method which comprises harmonized category indicators at the midpoint and the endpoint level. *In:* GOEDKOOP, M., HEIJUNGS, R., HUIJBREGTS, M., DE SCHRYVER, A., STRUIJS, J. & VAN ZELM, R. (eds.) *Report I: Characterization, first ed.*

REHBERG, M. J., ANDREWS, R. D., SWAIN, U. G. & CALKINS, D. G. 2009. Foraging behavior of adult female Steller sea lions during the breeding season in Southeast Alaska. *Marine Mammal Science,* 25**,** 588-604.

RISCH, D., CASTELLOTE, M., CLARK, C. W., DAVIS, G. E., DUGAN, P. J., HODGE, L. E., KUMAR, A., LUCKE, K., MELLINGER, D. K., NIEUKIRK, S. L., POPESCU, C. M., RAMP, C., READ, A. J., RICE, A. N., SILVA, M. A., SIEBERT, U., STAFFORD, K. M., VERDAAT, H. & VAN PARIJS, S. M. 2014. Seasonal migrations of North Atlantic minke whales: novel insights from large-scale passive acoustic monitoring networks. *Movement Ecology,* 2.

RISCH, D., WILSON, S. C., HOOGERWERF, M., VAN GEEL, N. C. F., EDWARDS, E. W. J. & BROOKES, K. L. 2019. Seasonal and diel acoustic presence of North Atlantic minke whales in the North Sea. *Scientific reports,* 9**,** 3571-3571.

ROBINSON, K. P., TETLEY, M. J. & MITCHELSON-JACOB, E. G. 2009. The distribution and habitat preference of coastally occurring minke whales (Balaenoptera acutorostrata) in the outer southern Moray Firth, northeast Scotland. *Journal of Coastal Conservation,* 13**,** 39-48.

ROCHMAN, C. M., BROWNE, M. A., UNDERWOOD, A. J., VAN FRANEKER, J. A., THOMPSON, RICHARD C. & AMARAL-ZETTLER, L. A. 2016. The ecological impacts of marine debris: unraveling the demonstrated evidence from what is perceived. *Ecology,* 97**,** 302-312.

RODRÍGUEZ, B., BÉCARES, J., RODRÍGUEZ, A. & ARCOS, J. M. 2013. Incidence of entanglements with marine debris by northern gannets (Morus bassanus) in the non-breeding grounds. *Marine Pollution Bulletin,* 75**,** 259-263.

ROSEL, P. E., HANSEN, L. & HOHN, A. A. 2009. Restricted dispersal in a continuously distributed marine species: common bottlenose dolphins *Tursiops truncatus* in coastal waters of the western North Atlantic. *Molecular Ecology,* 18**,** 5030-5045.

RUSSELL, D. J. F., MCCONNELL, B., THOMPSON, D., DUCK, C., MORRIS, C., HARWOOD, J. & MATTHIOPOULOS, J. 2013. Uncovering the links between foraging and breeding regions in a highly mobile mammal. *Journal of Applied Ecology,* 50**,** 499-509.

RYAN, P. G. 2015. Does size and buoyancy affect the long-distance transport of floating debris? *Environmental Research Letters,* 10**,** 084019.

RYBERG, M. W., HAUSCHILD, M. Z., WANG, F., AVEROUS-MONNERY, S. & LAURENT, A. 2019. Global environmental losses of plastics across their value chains. *Resources, Conservation and Recycling,* 151**,** 104459.

SAIC & CURRAN, M. A. 2006. *Life-cycle assessment: principles and practice,* Cincinnati, Ohio, National Risk Management Research Laboratory, Office of Research and and Development, US Environmental Protection Agency.

SALTON, M., KIRKWOOD, R., SLIP, D. & HARCOURT, R. 2019. Mechanisms for sex-based segregation in foraging behaviour by a polygynous marine carnivore. *Marine Ecology Progress Series,* 624**,** 213-226.

SANCHO, G., PUENTE, E., BILBAO, A., GOMEZ, E. & ARREGI, L. 2003. Catch rates of monkfish (Lophius spp.) by lost tangle nets in the Cantabrian Sea (northern Spain). *Fisheries Research,* 64**,** 129-139.

SAYER, S., ALLEN, R., HAWKES, L., HOCKLEY, K., JARVIS, D. & WITT, M. 2019. Pinnipeds, people and photo identification: The implications of grey seal movements for effective management of the species. *Journal of the Marine Biological Association of the United Kingdom***,** 1-10.

SCHREY, E. & VAUK, G. J. M. 1987. Records of entangled gannets (Sula bassana) at Helgoland, German Bight. *Marine Pollution Bulletin,* 18**,** 350-352.

SCHWARZ, A. E., LIGTHART, T. N., BOUKRIS, E. & VAN HARMELEN, T. 2019. Sources, transport, and accumulation of different types of plastic litter in aquatic environments: A review study. *Marine Pollution Bulletin,* 143**,** 92-100.

SEMEYN, D. J., CUSH, C. C., SCOLARDI, K. M., HEBERT, J., MCBRIDE, J. D., GREALISH, D. & REYNOLDS, J. E. 2011. Aerial surveys of manatees (Trichechus manatus) in Lee County, Florida, provide insights regarding manatee abundance and real time information for managers and enforcement officers. *Journal of Coastal Conservation,* 15**,** 573-583.

SHAUGHNESSY, P., MCKENZIE, J., LANCASTER, M., GOLDSWORTHY, S. & DENNIS, T. 2010. Australian fur seals establish haulout sites and breeding colony in South Australia. *Australian Journal of Zoology,* 58**,** 94-103.

SHOMURA, R. S. & YOSHIDA, H. O. Proceedings of the Workshop on the Fate and Impact of Marine Debris. 26-29 November 1984 1985 Honolulu, Hawaii. U.S. Dept. of Commerce. NOAA Technical Memorandum NMFSSWFC-54, 580.

SJÖBERG, M. & BALL, J. 2000. Grey seal, Halichoerus grypus, habitat selection around haulout sites in the Baltic Sea: bathymetry or central-place foraging? *Canadian Journal of Zoology,* 78**,** 1661-1667.

SJÖBERG, M., FEDAK, M. A. & MCCONNELL, B. J. 1995. Movements and diurnal behaviour patterns in a Baltic grey seal (Halichoerus grypus). *Polar Biology,* 15**,** 593-595.

SKAUG, H., ØIEN, N., SCHWEDER, T. & BØTHUN, G. 2004. Abundance of minke whales (Balaenoptera acutorostrata) in the Northeastern Atlantic. *Canadian Journal of Fisheries and Aquatic Sciences - CAN J FISHERIES AQUAT SCI,* 61**,** 870-886.

SONNEMANN, G. & VALDIVIA, S. 2017. Medellin Declaration on Marine Litter in Life Cycle Assessment and Management: Facilitated by the Forum for Sustainability through Life Cycle Innovation (FSLCI) in close cooperation with La Red Iberoamericana de Ciclo de Vida (RICV) on Wednesday 14 of June 2017. *The International Journal of Life Cycle Assessment*.

SONNTAG, N. & HÜPPOP, O. 2005. Snacks from the depth: summer and winter diet of common guillemots *uria aalge* around the island of Helgoland. *Atlantic Seabirds,* 7**,** 1-14.

STANILAND, I., BOYD, I. & REID, K. 2007. An energy–distance trade-off in a central-place forager, the Antarctic fur seal (Arctocephalus gazella). *Marine Biology,* 152**,** 233-241.

STANILAND, I. J., ROBINSON, S. L., SILK, J. R. D., WARREN, N. & TRATHAN, P. N. 2012. Winter distribution and haul-out behaviour of female Antarctic fur seals from South Georgia. *Marine Biology,* 159**,** 291-301.

TASKER, M. L., CAMPHUYSEN, C. J., COOPER, J., GARTHE, S., MONTEVECCHI, W. A. & BLABER, S. J. M. 2000. The impacts of fishing on marine birds. *ICES Journal of Marine Science,* 57**,** 531-547.

TER HALLE, A., JEANNEAU, L., MARTIGNAC, M., JARDÉ, E., PEDRONO, B., BRACH, L. & GIGAULT, J. 2017. Nanoplastic in the North Atlantic Subtropical Gyre. *Environmental Science & Technology,* 51**,** 13689-13697.

TER HALLE, A., LADIRAT, L., GENDRE, X., GOUDOUNECHE, D., PUSINERI, C., ROUTABOUL, C., TENAILLEAU, C., DUPLOYER, B. & PEREZ, E. 2016. Understanding the Fragmentation Pattern of Marine Plastic Debris. *Environmental Science & Technology,* 50**,** 5668-5675.

THAXTER, C. B., LASCELLES, B., SUGAR, K., COOK, A. S. C. P., ROOS, S., BOLTON, M., LANGSTON, R. H. W. & BURTON, N. H. K. 2012. Seabird foraging ranges as a preliminary tool for identifying candidate Marine Protected Areas. *Biological Conservation,* 156**,** 53-61.

THIEL, M., LUNA-JORQUERA, G., ÁLVAREZ-VARAS, R., GALLARDO, C., HINOJOSA, I. A., LUNA, N., MIRANDA-URBINA, D., MORALES, N., ORY, N., PACHECO, A. S., PORTFLITT-TORO, M. & ZAVALAGA, C. 2018. Impacts of Marine Plastic Pollution From Continental Coasts to Subtropical Gyres—Fish, Seabirds, and Other Vertebrates in the SE Pacific. *Frontiers in Marine Science,* 5.

THOMPSON, D. L., OVENDEN, T. S., PENNYCOTT, T. & NAGER, R. G. 2020. The prevalence and source of plastic incorporated into nests of five seabird species on a small offshore island. *Marine Pollution Bulletin,* 154**,** 111076.

TILZEY, R. D. J., GOLDSWORTHY, S. D., CAWTHORN, M., CALVERT, N., HAMER, D. J., RUSSELL, S., SHAUGHNESSY, P. D., WIZE, B. & STEWARDSON, C. 2006. Assessment of seal-fishery interactions in the winter blue grenadier fishery off west Tasmania and the development of fishing practices and Seal Exclusion Devices to mitigate seal bycatch by factory trawlers. Canberra, Australia: Fisheries Research and Development Corporation.

TORRES, L. G., ROSEL, P. E., D'AGROSA, C. & READ, A. J. 2003. IMPROVING MANAGEMENT OF OVERLAPPING BOTTLENOSE DOLPHIN ECOTYPES THROUGH SPATIAL ANALYSIS AND GENETICS. *Marine Mammal Science,* 19**,** 502-514.

TRAAS, T. P. & VAN LEEUWEN, C. J. 2007. Ecotoxicological Effects. *In:* VAN LEEUWEN, C. J. & VERMEIRE, T. G. (eds.) *Risk Assessment of Chemicals: An Introduction.* Netherlands: Dordrecht: Springer

TRANQUILLA, L. A. M., MONTEVECCHI, W. A., FIFIELD, D. A., HEDD, A., GASTON, A. J., ROBERTSON, G. J. & PHILLIPS, R. A. 2014. Individual winter movement strategies in two species of murre (Uria spp.) in the Northwest Atlantic. *PloS one,* 9**,** e90583-e90583.

UN ENVIRONMENT 2018. Mapping of global plastic value chain and plastic losses to the environment: with a particular focus on marine environment. *In:* RYBERG, M. W., LAURENT, A. & HAUSCHILD, M. Z. (eds.). Nairobi, Kenya: United Nations Environment Programme.

UNGER, B., HERR, H., BENKE, H., BÖHMERT, M., BURKHARDT-HOLM, P., DÄHNE, M., HILLMANN, M., WOLFF-SCHMIDT, K., WOHLSEIN, P. & SIEBERT, U. 2017. Marine debris in harbour porpoises and seals from German waters. *Mar Environ Res,* 130**,** 77-84.

URIAN, K. W., HOFMANN, S., WELLS, R. S. & READ, A. J. 2009. Fine-scale population structure of bottlenose dolphins (Tursiops truncatus) in Tampa Bay, Florida. *Marine Mammal Science,* 25**,** 619-638.

VAN DE MEENT, D. & HUIJBREGTS, M. A. J. 2005. Calculating life-cycle assessment effect factors from potentially affected fraction-based ecotoxicological response functions. *Environmental Toxicology and Chemistry,* 24**,** 1573-1578.

VAN DER HOOP, J., CORKERON, P. & MOORE, M. 2017. Entanglement is a costly life-history stage in large whales. *Ecology and Evolution,* 7**,** 92-106.

VAN FRANEKER, J. A., BLAIZE, C., DANIELSEN, J., FAIRCLOUGH, K., GOLLAN, J., GUSE, N., HANSEN, P.-L., HEUBECK, M., JENSEN, J.-K., LE GUILLOU, G., OLSEN, B., OLSEN, K.-O., PEDERSEN, J., STIENEN, E. W. M. & TURNER, D. M. 2011. Monitoring plastic ingestion by the northern fulmar Fulmarus glacialis in the North Sea. *Environmental Pollution,* 159**,** 2609-2615.

VAN FRANEKER, J. A. & LAW, K. L. 2015. Seabirds, gyres and global trends in plastic pollution. *Environmental Pollution,* 203**,** 89-96.

VAN WAEREBEEK, K., ANDRÉ, M., SEQUEIRA, M., MARTÍN, V., ROBINEAU, D., COLLET, A., PAPASTAVROU, V. & NDIAYE, E. 1999. Spatial and temporal distribution of the minke whale Balaenoptera acutorostrata (Lacepede, 1804), in the southern northeast Atlantic Ocean and the Mediterranean Sea, with reference to stock identity. . *J. Cetacean Res. Manage.,* 1**,** 223–237.

VERONES, F., BARE, J., BULLE, C., FRISCHKNECHT, R., HAUSCHILD, M., HELLWEG, S., HENDERSON, A., JOLLIET, O., LAURENT, A., LIAO, X., LINDNER, J. P., MAIA DE SOUZA, D., MICHELSEN, O., PATOUILLARD, L., PFISTER, S., POSTHUMA, L., PRADO, V., RIDOUTT, B., ROSENBAUM, R. K., SALA, S., UGAYA, C., VIEIRA, M. & FANTKE, P. 2017. LCIA framework and cross-cutting issues guidance within the UNEP-SETAC Life Cycle Initiative. *Journal of Cleaner Production,* 161**,** 957-967.

VINCENT, C., FEDAK, M. A., MCCONNELL, B. J., MEYNIER, L., SAINT-JEAN, C. & RIDOUX, V. 2005. Status and conservation of the grey seal, Halichœrus grypus, in France. *Biological Conservation,* 126**,** 62-73.

VINCENT, C., HUON, M., CAURANT, F., DABIN, W., DENIAU, A., DIXNEUF, S., DUPUIS, L., ELDER, J.-F., FREMAU, M.-H., HASSANI, S., HEMON, A., KARPOUZOPOULOS, J., LEFEUVRE, C., MCCONNELL, B., MOSS, S., PROVOST, P., SPITZ, J., TURPIN, Y. & RIDOUX, V. 2017. Grey and harbour seals in France: Distribution at sea, connectivity and trends in abundance at haulout sites. *Deep Sea Research Part II: Topical Studies in Oceanography*.

VKM, RUENESS, E. K., ASMYHR, M. G., KAUSRUD, K., NILSEN, E. B., ØIEN, N., DE BOER, H., ELDEGARD, K., HINDAR, K., HOLE, L. R., JÄRNEGREN, J., KIRKENDALL, L., MÅREN, I. E., NIELSEN, A., THORSTAD, E. B. & VELLE, G. 2019. A CITES risk assessment for the common minke whale (Balaneoptera acutorostrata). Opinion of the Panel on Alien Organisms and Trade in Endangered Species (CITES) of the Norwegian Scientific Committee for Food and Environment. . *VKM report 2019: 10.* Oslo, Norway: Norwegian Scientific Committee for Food and Environment (VKM).

VOLLMER, N. L. & ROSEL, P. E. 2013. A Review of Common Bottlenose Dolphins (<i>Tursiops truncatus truncatus</i>) in the Northern Gulf of Mexico: Population Biology, Potential Threats, and Management. *Southeastern Naturalist,* 12**,** 1-43, 43.

VOTIER, S. C., ARCHIBALD, K., MORGAN, G. & MORGAN, L. 2011. The use of plastic debris as nesting material by a colonial seabird and associated entanglement mortality. *Marine Pollution Bulletin,* 62**,** 168-172.

WAGGITT, J. J., EVANS, P. G. H., ANDRADE, J., BANKS, A. N., BOISSEAU, O., BOLTON, M., BRADBURY, G., BRERETON, T., CAMPHUYSEN, C. J., DURINCK, J., FELCE, T., FIJN, R. C., GARCIA-BARON, I., GARTHE, S., GEELHOED, S. C. V., GILLES, A., GOODALL, M., HAELTERS, J., HAMILTON, S., HARTNY-MILLS, L., HODGINS, N., JAMES, K., JESSOPP, M., KAVANAGH, A. S., LEOPOLD, M., LOHRENGEL, K., LOUZAO, M., MARKONES, N., MARTÍNEZ-CEDEIRA, J., Ó CADHLA, O., PERRY, S. L., PIERCE, G. J., RIDOUX, V., ROBINSON, K. P., SANTOS, M. B., SAAVEDRA, C., SKOV, H., STIENEN, E. W. M., SVEEGAARD, S., THOMPSON, P., VANERMEN, N., WALL, D., WEBB, A., WILSON, J., WANLESS, S. & HIDDINK, J. G. 2020. Distribution maps of cetacean and seabird populations in the North-East Atlantic. *Journal of Applied Ecology,* 57**,** 253-269.

WAKEFIELD, E., BODEY, T., BEARHOP, S., BLACKBURN, J., COLHOUN, K., DAVIES, R., DWYER, R., GREEN, J., GRÉMILLET, D., JACKSON, A., JESSOPP, M., KANE, A.,

LANGSTON, R., LESCROEL, A., MURRAY, S., NUZ, M., PATRICK, S., PÉRON, C., SOANES, L. & HAMER, K. 2013. Space Partitioning Without Territoriality in Gannets. *Science (New York, N.Y.),* 341.

WALUDA, C., GREGORY, S. & DUNN, M. 2010. Long-term variability in the abundance of Antarctic fur seals Arctocephalus gazella at Signy Island, South Orkneys. *Polar Biology,* 33**,** 305-312.

WALUDA, C., STANILAND, I., DUNN, M., THORPE, S., GRILLY, E., WHITELAW, M. & HUGHES, K. 2020. Thirty years of marine debris in the Southern Ocean: Annual surveys of two island shores in the Scotia Sea. *Environment international,* 136**,** 105460.

WALUDA, C. M. & STANILAND, I. J. 2013. Entanglement of Antarctic fur seals at Bird Island, South Georgia. *Mar Pollut Bull,* 74**,** 244-52.

WANLESS, S. 2002. *Northern Gannet (Gannet) Morus bassanus.,* London, Poyser.

WEIGLE, B. L., WRIGHT, I. E., ROSS, M. & FLAMM, R. O. 2001. Movements of radio-tagged manatees in Tampa Bay and along Florida's west coast, 1991–1996. *Florida Marine Research InstitutecTechnical Report.* St. Petersburg, FL.

WEIMERSKIRCH, H., CHASTEL, O., CHEREL, Y., HENDEN, J.-A. & TVERAA, T. 2001. Nest attendance and foraging movements of northern fulmars rearing chicks at Bj??rn??ya Barents Sea. *Polar Biology,* 24**,** 83-88.

WELLS, R. S., NATOLI, A. & BRAULIK, G. 2019. *Tursiops truncatus* (errata version published in 2019). *The IUCN Red List of Threatened Species,* 2019.

WELLS, R. S., RHINEHART, H. L., CUNNINGHAM, P., WHALEY, J., BARAN, M., KOBERNA, C. & COSTA, D. P. 1999. Long distance offshore movements of bottlenose dolphins. *Marine Mammal Science,* 15**,** 1098-1114.

WERNER, S., BUDZIAK, A., VAN FRANEKER, J., GALGANI, F., HANKE, G., MAES, T., MATIDDI, M., NILSSON, P., OOSTERBAAN, L., PRIESTLAND, E., THOMPSON, R., MIRA VEIGA, J. & VLACHOGIANNI, T. 2016. Harm caused by Marine Litter. Luxembourg: MSFD GES Technical Group on Marine Litter.

WILCOX, C., MALLOS, N. J., LEONARD, G. H., RODRIGUEZ, A. & HARDESTY, B. D. 2016. Using expert elicitation to estimate the impacts of plastic pollution on marine wildlife. *Marine Policy,* 65**,** 107-114.

WILLIAMS, R., GERO, S., BEJDER, L., CALAMBOKIDIS, J., KRAUS, S. D., LUSSEAU, D., READ, A. J. & ROBBINS, J. 2011. Underestimating the damage: interpreting cetacean carcass recoveries in the context of the Deepwater Horizon/BP incident. *Conservation Letters,* 4**,** 228-233.

WOMBLE, J. N., SIGLER, M. F. & WILLSON, M. F. 2009. Linking seasonal distribution patterns with prey availability in a central-place forager, the Steller sea lion. *Journal of Biogeography,* 36**,** 439-451.

WOODS, J. S., RØDDER, G. & VERONES, F. 2019. An effect factor approach for quantifying the entanglement impact on marine species of macroplastic debris within life cycle impact assessment. *Ecological Indicators,* 99**,** 61-66.

WOODS, J. S., VELTMAN, K., HUIJBREGTS, M., VERONES, F. & HERTWICH, E. 2016. Towards a meaningful assessment of marine ecological impacts in life cycle assessment (LCA). *Environment International,* 89-90**,** 48-61.

Appendices

Appendix 1: Confidence level criteria

Appendix 2: Extended potential exposure area descriptions

Appendix 3: Estimated plastic debris concentrations in mean and extended exposure area using mean and maximum summary statistics, and associated EC5 values.

Appendix 1: Confidence level criteria

A high confidence level was assigned to delineated mean potential exposure areas that fulfilled all criteria of class 1 (Table 3). For mammals that are not associated with terrestrial sites, a fulfilment of the criteria of class 2 was sufficient to obtain high confidence level. As the purpose of obtaining the mean ranges was to delineate population specific areas, ranges based on data on a class 3. spatial level were assigned low confidence, regardless of fulfillment of the other criteria. All other combinations of classes were assigned a moderate confidence level. A moderate confidence level was also assigned when there were discrepancies in the literature, or distinct geographic boundaries between the assumed entangled population and other populations of the same species could not be ascertained. Data where the method was unknown (class 3) were not included as basis for delineating potential exposure areas.

Table 3: Robustness criteria serving as the basis for the assigning of confidence levels to mean potential exposure areas.

Class	Study method	Spatial level	Population sample match
1.	Direct[A]	Colony specific	Representative of population
2.	Observational[B] or indirect[C]	Region specific	Missing data on 1 age/sex class
3.	Unknown	Species general	Missing data on >1 age/sex class

[A]: Tracking of individuals, e.g. satellite devices, GPS tags. [B]: Visual and/or acoustic records. [C]: Range derived based on time away from colony and estimated flight speed as proxy

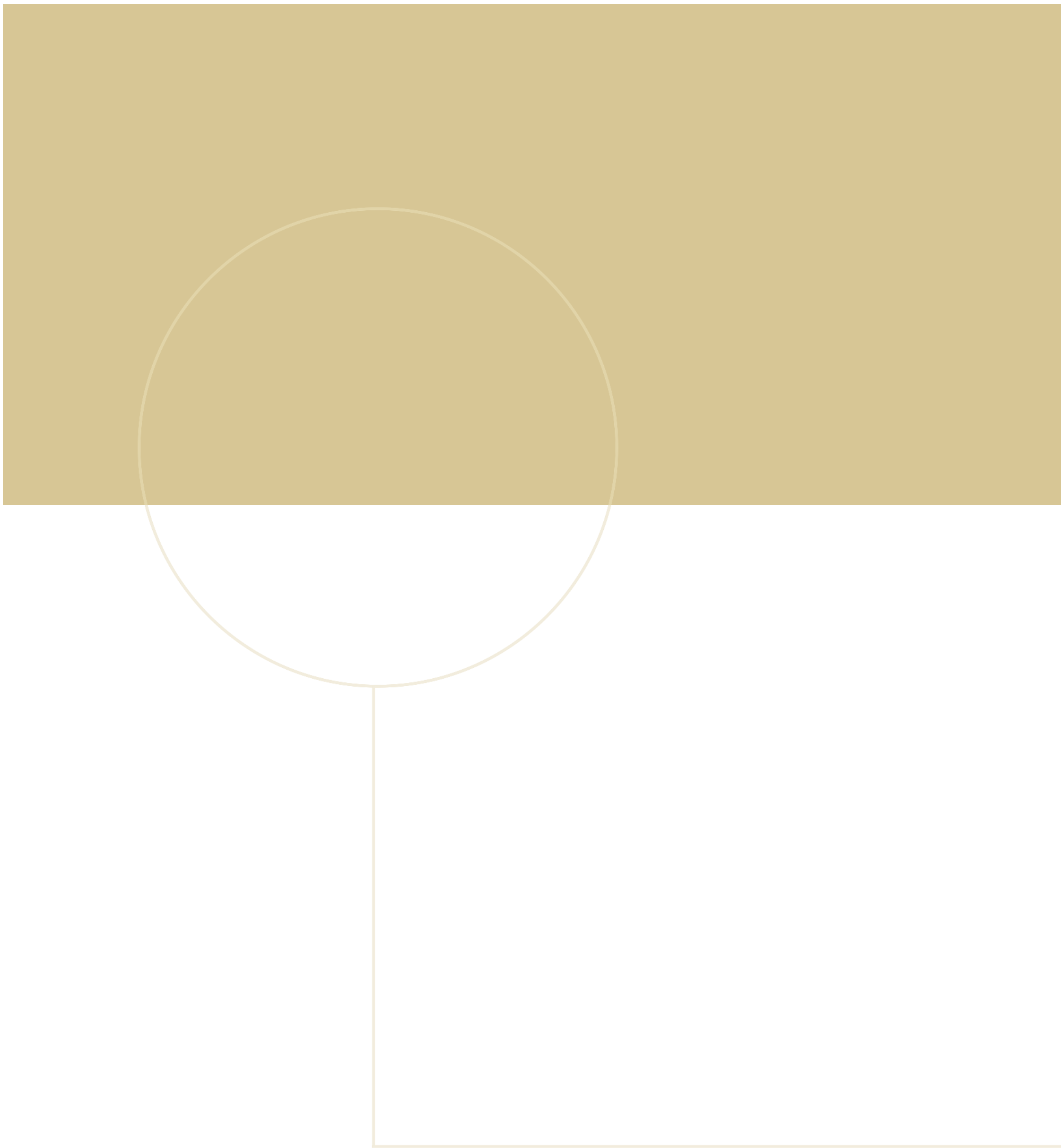
Appendix 2: Extended potential exposure area descriptions applied in sensitivity testing for species without in-text defined maximum ranges used as extended potential exposure areas.

Species	**Extended range**
Australian Fur Seal (A. p. doriferus)	Area with depth of 0-200 m, cut in west and east by IUCN subspecies range boundaries
Common bottlenose dolphin (*T. truncatus*)	Full IUCN distribution
Florida Manatee (*T. m. latirostris*)	IUCN distribution within the United States
Grey seal (*H. grypus*)	IUCN distribution in Eastern Atlantic (excluding Iceland)
North Atlantic right whale (*E. glacialis*)	Full IUCN distribution
Steller Sea Lion *(E. j. monteriensis*)	IUCN distribution of subspecies

Appendix 3: Estimated maximum plastic debris concentrations (g/km^2) in mean potential exposure area, and estimated mean and maximum plastic concentrations in extended potential exposure area, with associated EC5 values derived from plastic debris concentrations and entanglement rates using standard linear regression.

Species	Entanglement rate (%)	**Mean potential exposure area**		**Extended potential exposure area**			
		max plastic conc. (g/km^2)	EC5	Mean plastic conc. (g/km^2)	EC5	Max plastic conc. (g/km^2)	EC5
M. bassanus	-	-	3,48E+03	-	1,39E+03	-	1,08E+04
Breeding	3.05	9,93E+02	1,63E+03	1,25E+03	2,05E+03	6,97E+03	1,14E+04
Wintering	6.88*	-	6,25E+03	5,49E+02	3,99E+02	1,36E+04	9,86E+03
Cantabrian sea	0.11	8,37E+03	-	-	-	-	-
Gulf of Cadiz	0.36	1,69E+03	-	-	-	-	-
Mauritania	20.16	3,26E+03	-	-	-	-	-
U. aalge	1.05	9,93E+02	4,73E+03	9,02E+02	4,30E+03	9,93E+02	4,73E+03
F. glacialis	1.80	1,24E+03	3,44E+03	9,38E+02	2,61E+03	1,19E+04	3,32E+04
A. gazella	0.04*	-	2,75E+02	6,03E-01	7,18E+01	5,92E+01	7,05E+03
South Georgia	0.05	3,02E+00	-	-	-	-	-
South Orkney	0.02	5,30E-01	-	-	-	-	-
A. p. doriferus	0.014	1,04E+04	3,73E+06	2,95E+03	1,05E+06	3,11E+04	1,11E+07
T. truncatus	0.08	6,31E+04	4,05E+06	8,72E+02	5,59E+04	5,54E+05	3,55E+07
T. m. latirostris	0.46	6,31E+04	6,86E+05	4,99E+03	5,42E+04	6,31E+04	6,86E+05
H. grypus	4.30	1,86E+03	2,16E+03	6,20E+02	7,21E+02	4,56E+04	5,31E+04
E. glacialis	1.88	4,36E+04	1,16E+05	6,61E+02	1,76E+03	6,44E+04	1,72E+05
E. j. monteriensis	0.26	9,49E+03	1,82E+05	9,70E+01	1,87E+03	9,49E+03	1,82E+05

* Entanglement rate averaged from all given entanglement rates used to derive EC5 only when applying extended potential exposure area.



NTNU
Norwegian University of
Science and Technology