Ecological Indicators 135 (2022) 108509

Contents lists available at ScienceDirect

Ecological Indicators

journal homepage: www.elsevier.com/locate/ecolind

Original Articles

Global distribution of potential impact hotspots for marine plastic debris entanglement

Marthe A. Høiberg *, John S. Woods, Francesca Verones

Industrial Ecology Programme, Department of Energy and Process Engineering, NTNU, Trondheim, Norway

ARTICLE INFO

Keywords:
Entanglement
Plastic waste
Macroplastic
Ghost gear
Marine biodiversity
Impact assessment

ABSTRACT

Marine animals have been known to interact with and become entangled in plastic debris for decades. Despite increasing annual input volumes of plastic waste to the natural environment and the threat this constitutes to marine biodiversity, impacts of mismanaged plastic waste generally remain unquantified in environmental impact assessments. In this paper, we develop a Species Sensitivity Distribution (SSD) approach for estimating a spatially differentiated indicator of potential macroplastic entanglement impacts with global coverage of the world's oceans. This constitutes a key modelling step that contributes towards the inclusion of plastic litter effects in impact assessments. We gathered entanglement incidence data for 20 species of marine mammals, turtles and birds from different populations and marine regions. To capture species-specific sensitivities to entanglement and spatially varying concentrations of plastic debris, concentration–response modelling of field data was used to develop the SSD-based model. This was achieved by linking population specific entanglement records to corresponding regional areas of exposure and an existing global model of plastic debris concentrations. The SSD was further applied to derive an estimate of the Potentially Affected Fraction (PAF) of species on a global scale, highlighting regional hotspots of potential entanglement impacts at current levels of marine plastic pollution. This indicator can be adapted and applied in impact assessments in order to account for potential impacts of mismanaged plastic waste ending up in marine ecosystems.

1. Introduction

Marine plastic pollution is recognized as a globally pervasive environmental issue (Villarrubia-Gómez et al., 2018) associated with a range of ecological, as well as social and economic costs (Beaumont et al., 2019). Yet, annual inputs of plastic waste to landfills and the natural environment are expected to continue increasing (Lebreton & Andrady, 2019). According to two recent modeling studies, not even best-case scenarios for current management strategies to mitigate plastic pollution are sufficient to offset the increasing plastic waste accumulation (Borrelle et al., 2020; Lau et al., 2020). Littered or inadequately disposed plastic items carried by wind, rivers (Schmidt et al., 2017) or dumped directly at sea (Ryan, 2015) are distributed throughout the marine environment, accumulating in zones such as oceanic gyres (Lebreton et al., 2018), along coastlines and on the deep sea floor (Booth et al., 2017). Owing to the unknown timescales of complete degradation of synthetic polymers in the marine environment, the oceans virtually function as a sink for plastic debris (Law, 2017).

Negative impacts of plastic debris on marine mammals, seabirds and sea turtles have been observed for decades (Shomura & Yoshida, 1985). Particularly, the lethal potential of macroplastic entanglements for larger bodied marine animals is widely reported (Gall & Thompson, 2015; Werner et al., 2016; Wilcox et al., 2016). Over 350 unique species have now been registered as observed entangled in plastic debris (Kühn & van Franeker, 2020). Entanglement was defined by Laist (1997) as an interaction between marine life and anthropogenic debris that entraps animals or entangles their appendages by 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 can cause lacerations, infections and subsequent mortality (Dolman & Moore, 2017). Designed to be durable and catch marine animals, abandoned, lost or discarded fishing gear (ALDFG) made of plastic is frequently identified as the primary source of entanglements for many marine species (Duncan et al., 2017).

Not only do entanglements entail welfare implications for the individual (Byard & Machado, 2019), they also constitute a threat to marine biodiversity (Galgani et al., 2013) by potentially exacerbating declines in populations of vulnerable species (Perez-Venegas et al 2021). The

* Corresponding author.
E-mail address: marthe.a.hoiberg@ntnu.no (M.A. Høiberg).

https://doi.org/10.1016/j.ecolind.2021.108509
Received 1 November 2021; Received in revised form 22 December 2021; Accepted 24 December 2021
Available online 29 December 2021

endangered Hawaiian monk seal (*Neomonachus schauinslandi*), for example, has one of the highest documented entanglement rates (Antonelis et al., 2006), which combined with other stressors and a low genetic diversity threaten this small population (Nelms et al., 2021). In general, it is a difficult and costly endeavor to systematically observe and record wide ranging marine fauna (Martins et al., 2019; Wilcox et al., 2016), and the prevalence of injuries and mortalities caused by entanglements is thus likely underestimated. As quantitative assessments are also limited (O'Hanlon et al., 2019), global trends in entanglement impacts remain a knowledge gap (Nelms et al., 2021). This makes it a challenge to account for the impacts of plastic debris in environmental assessments in a meaningful manner. Leaving the negative effects of mismanaged plastic on marine ecosystems unquantified in such assessments implies accepting an obviously flawed assumption of a 100% waste collection rate, where all plastic goes to recycling, incineration or landfill (Boucher et al., 2019).

Attempts at estimating entanglement rates within species populations have been made using different approaches. For example, some authors report the annual share of entangled individuals obtained by multi-year observations of the same populations (Waluda & Staniland, 2013), while others use stranding databases and refer to a cumulative number of individuals found entangled over a larger and less defined geographic area (Adimey et al., 2014). Although this results in a data foundation that is varying across species populations and regions, it is possible to model the potential for entanglement impacts by collating the species-specific records and linking observed responses to environmental plastic debris concentrations. Quantifying an indicator of this impact can contribute towards highlighting the entanglement issue and gaining a more comprehensive evaluation of the consequences of current plastic consumption and waste generation volumes if applied to inform environmental assessments.

One approach to achieve this is through the use of Species Sensitivity Distribution (SSD) models, which are frequently applied for quantifying effects of various stressors on ecosystems (Hauschild & Huijbregts, 2015). SSDs are probabilistic models constructed based on dose–response modelling, which includes estimating the relationship between the exposure to a stressor and an observed effect. Moreover, it allows for an appreciation of interspecies variation in sensitivities to the same stressor (Posthuma et al., 2001), for example macroplastic debris entanglement. Although SSDs are traditionally constructed based on laboratory experiments, a handful of ecotoxicological studies have also utilized field monitoring data (e.g. Cormier and Suter, 2013; Leung et al., 2005) which by some are considered more ecologically relevant (Hoondert et al., 2018).

In this paper, we adopt a field-based SSD approach by combining observed entanglement rates from species populations with corresponding regional areas of exposure. Based on this, we present a spatially-differentiated indicator of current macroplastic entanglement impacts across species in terms of the Potentially Affected Fraction (PAF) of species. These results, which cover ocean surface waters globally, are then discussed in relation to locations of Marine Protected Areas (MPA) generally and in selected regions. It is particularly relevant to consider the potential for impacts within MPAs as they are established with the aim of achieving long-term conservation of nature, albeit to varying degrees (Day et al., 2019). Finally, the potential for application of the model presented in this paper in impact assessment methods is outlined.

2. Methodology

The approach to modelling current potential impact levels of entanglement consists of five steps (Fig. 1), each of which are elaborated in the subsequent sections.

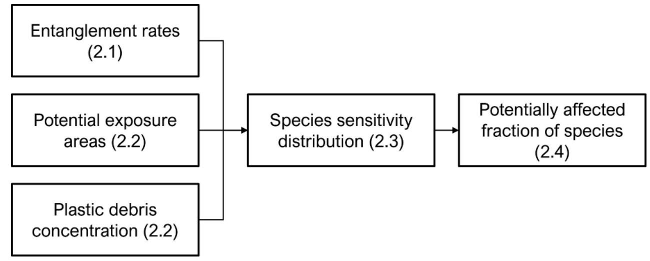

Fig. 1. Overview of modelling approach and components. Note that the delineation of potential exposure areas and estimation of plastic debris concentration within those areas are presented in one joint section (2.2).

2.1. Entanglement rates

Entanglement rates for marine species were collected from peer reviewed papers through the databases Web of Science and Google Scholar using search terms such as "entanglement", "marine" "plastic", "debris" and "litter". In addition, a Google search for reports from agencies that monitor plastic debris entanglements was also made using the same keywords. Records of entanglements dating back several decades exist in the literature, but a restriction to events occurring after the year 2000 was made as a compromise between age of data and achieving enough datapoints for the model. As entanglement has mainly been studied in large marine fauna, the current approach implicitly considers entanglement only by macroplastic, commonly defined as items with a diameter > 5 mm (GESAMP, 2019).

From a policymaking and management point of view, it can be important to distinguish between effects caused by plastic debris and ALDFG, and those caused by animals swimming into fishing gear operational at the time of the entanglement event. This distinction is, however, often not trivial to make in nature (Asmutis-Silvia et al., 2017) . In this paper, reports of entanglements that were explicitly identified as a consequence of active/stationary fishing gear were excluded, as we are relating the entanglement rates to environmental plastic debris concentrations. Applying the above-mentioned restrictions on the literature search led to a total of 22 independent sources of entanglement rates.

The rates were gathered both from reports of live observations of species populations, and stranding records in order to gain a larger representation of species and geographic areas in the dataset. 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, entanglement rates from stranding events and the live population have been found to be within the same order of magnitude (Allyn & Scordino, 2020). A difference between these two data types is that a smaller part of the population can be observed through stranding data. To account for this, entanglement rates from stranded animals were only included in the model when they were derived from records of > 100 individuals or from a period of at least five years. All entanglement rates in the model represent the annual share of observed entangled individuals relative to a given total number of individuals (Eq. (1)):

$$Annual\ \ entanglement\ \ rate = \frac{\frac{Entangled\ \ individuals}{Total\ \ individuals}}{Years\ \ of\ \ observation} \quad (1)$$

The current model contains rates normalized to a theoretical population estimate or an observed sample size (Table S1, Supporting Information (SI)), depending on what was reported in the primary source. In total, 15 of the rates are normalized to a population estimate while 25 are normalized to the reported sample size.

2.2. Potential exposure areas

Observed/estimated entanglement rates are species- and spatially specific and are the result of species behavior in the presence of plastic debris materials within a specific geographic area i.e., a potential exposure area. While existing species distribution maps from e.g., the International Union for Conservation of Nature (IUCN, 2021) allow for an understanding of the global spatial pattern of species, they have limited utility as core exposure areas for regional populations. In this modelling approach, spatially explicit population ranges are required to attain a more relevant link between concentrations and response, i.e., population specific geographic areas of exposure and observed entanglement rates. The potential exposure areas should thus reflect the areas that the populations utilize most frequently, assuming that this is also where they are most likely to encounter and get entangled in plastic debris. The geographical extent of these areas was determined based on the ecology of each (sub)species, and largely defined based on foraging ranges collected from observational or telemetry tracking studies of the specific populations. Foraging areas reflect where a population is more likely to encounter plastic debris at sea (Thiel et al., 2018), which can be explained by different feeding strategies and greater temporal exposure within these areas.

The mean foraging range of a population was used to define the exposure area for all entanglement rates gathered from observations on land-based colonies (bird cliffs or haul-out sites for seals). For species that are not colonial or central-place foragers, e.g., cetaceans such as the Minke whale (*Balaenoptera acutorostrata*), bathymetry data from GEBCO (2020) in combination with estimated abundance maps obtained from literature was used to delineate the relevant areas. For sea turtles, where entanglement rates could only be obtained from stranding data, a larger area needed to be assumed as they can drift from far distances to the locations they are registered at (e.g. Jensen et al., 2013). As such, predefined areas termed regional management units (RMU) based on expert knowledge on the different populations (Wallace et al., 2010) were directly applied for all sea turtle species. Although RMUs represent full regional ranges rather than foraging ranges, they are considered the best estimate for these wide-ranging populations and applying the RMUs ensures consistency across all turtle species. Further details on the rationale behind each population-specific potential exposure area can be found in the supporting information. The areas were mapped using ArcGIS® software by Esri (SI, Figure S1).

The spatial distribution and concentration of macroplastic debris was provided by Eriksen et al. (2014) as a global estimate of the mass of floating plastic debris (of sizes > 4.75 mm) per square kilometer (g/km^2) of the surface of the ocean. The estimates are given on a 0.2 decimal degrees grid-cell resolution and derived from an oceanographic dispersal model calibrated by data from sampling conducted during expeditions over the period 2007–2013. To link the entanglement rates to the plastic concentrations found within the geographical areas of the respective population, the mean macroplastic debris concentrations (kg/km^2) were calculated for each potential exposure area using the “Spatial analyst: zonal statistics” tool in ArcGIS.

2.3. Species sensitivity distribution (SSD) modelling

An SSD model uses a statistical distribution to describe the sensitivity of a selection of species intended to be representative for most species of interest (Posthuma et al., 2001). In order to estimate the parameters of an SSD, effect concentrations for multiple species at a standardized effect level, commonly chosen as 50% of the population affected (EC50), are required as input. This is achieved here by concentration–response modelling, utilizing the estimated entanglement rates and plastic debris concentration data to predict the effective concentrations.

The mean concentration (kg/km^2) of plastic debris (ECx_i) within the potential exposure areas in combination with the species population specific entanglement rates (x_i) were used to calculate linear concentration–response relationships for all species. A linear approach with a zero intercept is selected, as the relation between concentrations of macroplastic debris and observed entanglement effects is not empirically known, and this is common practice when background concentrations of a stressor in the environment is unknown (Hauschild & Huijbregts, 2015).

Our SSD model uses annual entanglement rates gathered from field data, where the majority are found to be below 5% (SI, Table S1). As such, EC5 was considered a more appropriate effect concentration for construction of the SSD in order to minimize extrapolation distances. The rate of entanglements sufficient to cause population level effects for a given species population is not empirically known. However, a recent scenario modelling study on a South American fur seal colony suggests that the rates reported in literature for pinnipeds, also applied in this model, can over time have significant effects on population-level dynamics because of decreased population growth rates (Perez-Venegas et al., 2021). In the following, we apply the 5% annual entanglement rate of populations as the threshold and definition for a species being affected by macroplastic debris entanglement. In the current model, “affected” includes both chronic entanglements and mortality. By assuming zero entanglement effects at a concentration of zero plastic debris, the EC5 for each species *i* is derived by dividing 5 (the response) by the estimated slope parameter *b* of the linear regression for every species *i*:

$$EC5_i = \frac{5}{b_i} \tag{2}$$

For some (5 out of 20) of the species in the current dataset, more than one entanglement rate was available for estimating the slope that makes it possible to extrapolate the species-specific EC5 (Eq. (2)). All the EC5 datapoints were then plotted in R using the package SSD tools (Thorley & Schwarz, 2018) to produce a log-logistic, cumulative distribution of the sensitivity of the different species. For later use in testing model sensitivity, the hazardous concentration at which 50% of the species are affected (HC50) was derived along with a confidence interval using parametric bootstrapping.

2.4. Potentially affected fraction of species (PAF)

The distribution function from the SSD curve with its associated parameter values (SI, Table S2) was applied to estimate a PAF value for every grid cell in the ocean, where P_i is the concentration (kg/km^2) of macroplastic in location *i*, α is the location parameter and the constant β is the curve’s scale (Eq.3):

$$PAF_i = \frac{1}{1 + e^{-(\ln P_i - \alpha)/\beta}} \tag{3}$$

PAF can take any value between 0 and 1 i.e., representing 0 to 100% of species affected at a given location depending on the plastic debris concentrations in that location. When referring to fraction of species affected and potential impact levels in the following, this is based on the sensitivities of the 20 species in the current model. The function (Eq. (3)) was plotted in ArcGIS, using the “Spatial analyst: raster calculator” tool, producing a global overview of the potentially affected fraction of species.

The distribution of the PAF was also evaluated on a smaller and defined scale by considering Marine Protected Areas (MPA) of all IUCN protected area categories, extracted from the World Database on Protected Areas (IUCN and UNEP-WCMC, 2021). The average PAF value within MPAs was derived using the “Spatial analyst: zonal statistics” tool in ArcGIS. It should be noted that the values within very small MPAs are

Table 1
Overview of species and their respective EC5 values, i.e., plastic debris concentration (kg/km^2) at which 5% of population is annually entangled, derived from estimated plastic debris exposure and reported entanglement rates (See SI, Table S1 for numbered reference list of entanglement rate sources).

Common name	Scientific name	EC5	Sources
Antarctic fur seal	*Arctocephalus gazella*	5.02E-02	[17–18]
Australian fur seal	*Arctocephalus pusillus doriferus*	2.07E+02	[24]
New Zealand fur seal	*Arctocephalus forsteri*	1.82E+01	[31] [60]
Northern fur seal	*Callorhinus ursinus*	2.89E+00	[63–64]
Australian sea lion	*Neophoca cinerea*	1.74E+01	[31]
Steller sea lion	*Eumetopias jubatus monteriensis*	1.34E+00	[53–54]
California sea lion	*Zalophus californianus*	2.53E-01	[54]
Grey seal	*Halichoerus grypus*	1.80E+00	[49]
Hawaiian monk seal	*Neomonachus schauinslandi*	4.89E+00	[57]
Florida manatee	*Trichechus manatus latirostris*	3.49E+01	[35]
Common bottlenose dolphin	*Tursiops truncatus*	5.85E+01	[35]
Common minke whale	*Balaenoptera acutorostrata*	4.48E+00	[38–39]
Hawksbill sea turtle	*Eretmochelys imbricata*	1.85E+01	[35]
Leatherback sea turtle	*Dermochelys coriacea*	1.25E+01	[35] [66–67]
Olive ridley sea turtle	*Lepidochelys olivacea*	1.89E+02	[68–69]
Kemp's ridley sea turtle	*Lepidochelys kempii*	2.86E+01	[35]
Loggerhead sea turtle	*Caretta caretta*	1.03E+01	[35] [66–67] [70–71]
Northern gannet	*Morus bassanus*	2.40E+00	[3] [6] [8]
Common guillemot	*Uria aalge*	4.65E+00	[3]
Northern fulmar	*Fulmarus glacialis*	3.84E+00	[3]

not accounted for as the zonal statistics tool cannot return values for areas that are smaller than the resolution of the plastic debris concentration dataset, i.e., 0.2 decimal degrees.

2.5. Sensitivity analysis

Modelled/estimated parameters that may influence the output and that were feasible to assess in the current work include:

i) Entanglement rates
ii) Plastic debris concentrations in potential exposure areas
iii) The size and extent of potential exposure areas

Although the main model output is the global distribution of PAF, the sensitivity was tested by examining the output at the EC5- and SSD-level, as these are directly used to relate plastic debris concentrations to a PAF value. The sensitivity of the model to entanglement rates and plastic debris concentrations was tested by increasing these parameters separately for all species by 10%. In order to test the sensitivity of the extent of the potential exposure area, each area was increased by approximately 100% using the GIS tool BufferByPercentage. This tool extends the general area in every direction of a polygon, meaning that it may also extend onto land areas which are excluded from the model. As such, the increase does not represent an actual doubling of the area that is used to calculate the exposure concentrations. As some of the smaller exposure areas are at the size of one or two plastic debris concentration pixels in radius (Figure S1), it was necessary to apply a 100% increase in order to induce the inclusion of new pixel values for the calculation of average concentrations. The sensitivity to the change in exposure area sizes was assessed by comparing the SSD curves in terms of the HC50 value, the range of EC5 values and potential shifts in species placement on the curve.

3. Results and discussion

3.1. Species sensitivity distribution (SSD)

The literature search resulted in a dataset of 12 marine mammal species, 5 sea turtles and 3 seabird species (Table 1). Several of these species are often highlighted in the context of plastic debris impacts and frequently found in literature and reports on entanglements (e.g. Duncan et al., 2017; Nelms et al., 2021).

The EC5 values, which can be understood as sensitivities to plastic debris entanglement, vary by five orders of magnitude for the different species included in the model (Table 1). The underlying annual entanglement rates that were used to derive the EC5 values ranged from 0.02% recorded for the Antarctic fur seal *(A. gazella)* population on Signy Island (Waluda & Staniland, 2013) to 20.16% of northern gannets (*M. bassanus)* observed off the coast of Mauritania (Rodríguez et al., 2013). In addition to the population specific entanglement rates, the modelled sensitivity of a species is determined by the estimated plastic debris concentrations within the respective potential exposure areas (SI, Table S1). The computed EC5 values can be plotted on an SSD curve where the interspecies differences in sensitivity to the stressor can be easily observed (Fig. 2).

The loglogistic cumulative distribution function estimates that 50% of the species will be affected by entanglements at a plastic debris concentration of 7.8 kg/km^2 of ocean surface waters (Fig. 2). While the most sensitive species, the Antarctic fur seal (*A. gazella*), is estimated to be affected at already approximately 0.05 kg/km^2, the Australian fur seal *(A. p. doriferus*) is not affected until concentrations reach 206 kg/km^2. Noticeably, the reported entanglement rates for these two species are within the same order of magnitude (Hofmeyr et al., 2006; McIntosh et al., 2015; Waluda & Staniland, 2013) and relatively low (SI, Table S1). Despite both being fur seals, which involves sharing some ecological traits, they occupy different habitats which can explain their large difference in estimated sensitivities. In addition, *A. p. doriferus* forages in an area of Southern Australia with high plastic debris concentrations, while *A. gazella* is found in the Southern Ocean, which is associated with some of the lowest estimated plastic debris concentrations globally (Eriksen et al. 2014). As such, the coupling of a higher entanglement rate to a lower mean plastic concentration predicts that the species is more prone to becoming entangled.

While the different mammals are spread evenly throughout the curve, the group of marine birds is appearing on the lower end and all the sea turtle species are above the HC50 threshold (Fig. 2). Although

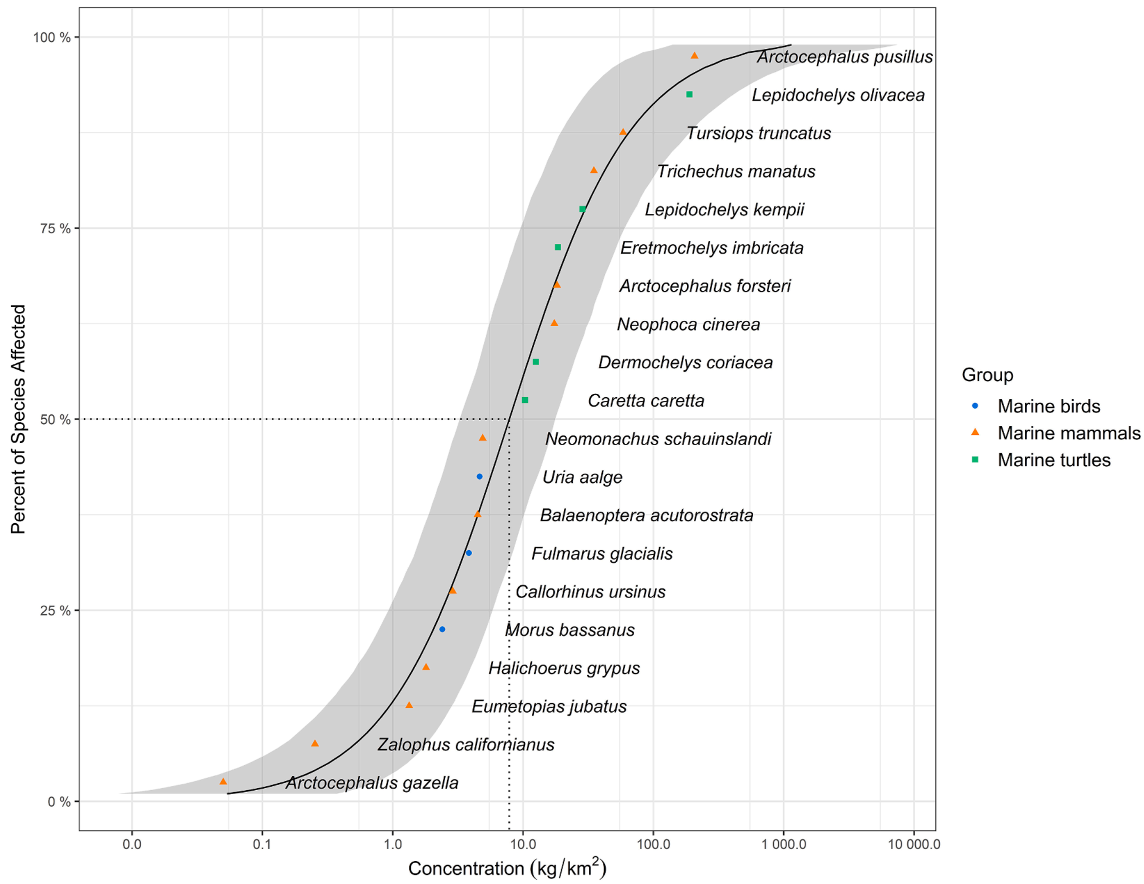

Fig. 2. Species sensitivity distribution (SSD) where each dot corresponds to the EC5 for a single species within the groups marine mammals (orange), birds (blue) and turtles (green), and the shaded grey area constitutes the 95% confidence interval. n = 20 species. Each EC5 datapoint with the underlying data can be found in the SI, Table S1. (For interpretation of the references to color in this figure legend, the reader is referred to the web version of this article.)

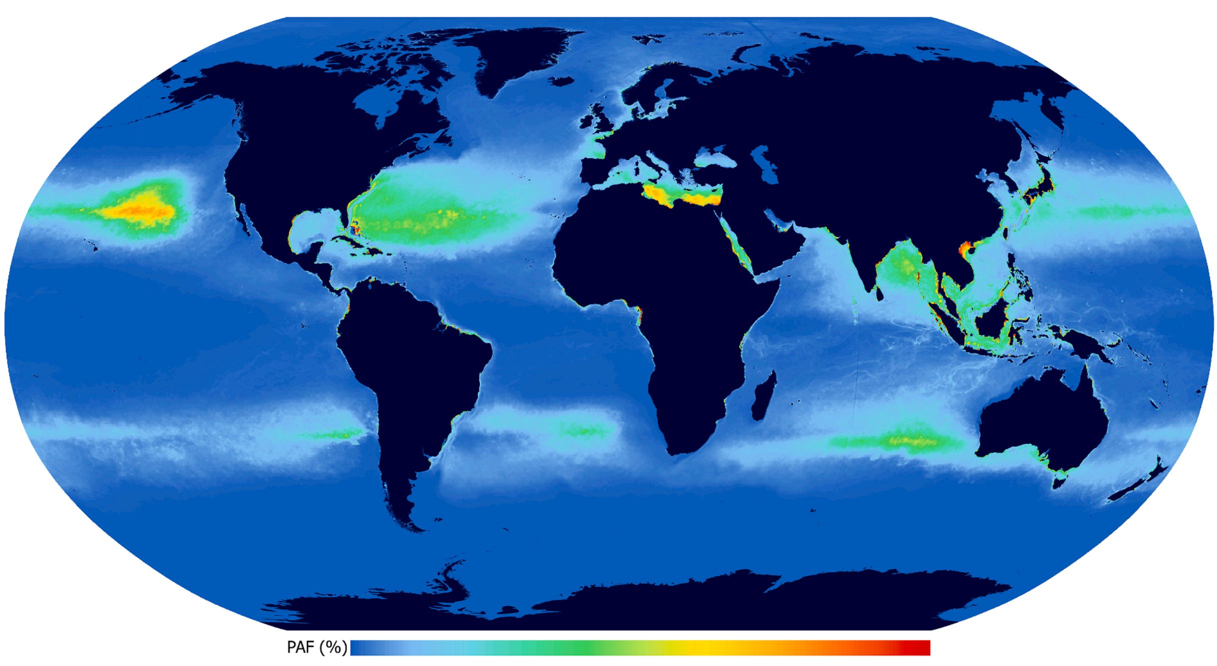

Fig. 3. Global distribution of potentially affected fraction (PAF) of species, based on a species sensitivity distribution (section 3.1) and estimates of current macroplastic debris concentrations from Eriksen et al (2014).

this could reflect potential differences in sensitivities to plastic debris for these groups, larger sample sizes and more species of particularly sea birds would be required in order to make a valid comparison.

3.2. Potentially affected fraction (PAF) of species

Based on the sensitivities of 20 species from all ocean basins, we derive a global and spatially explicit distribution of the potentially affected fraction (PAF) of species under current macroplastic debris concentration estimates (Fig. 3).

The current results (Fig. 3) suggest that across vast areas of the ocean, e.g., in the northern Pacific and Atlantic oceans, plastic debris concentrations in the surface waters are at a level where several species could potentially be affected. Compared to many other anthropogenic stressors, plastic debris reaches remote areas that are otherwise free of direct human impact, owing to its persistence and capability of being carried by wind and currents over great distances. This results in a pattern of higher estimated impact levels in areas known as accumulation zones for plastic debris, such as the large ocean gyres (Cózar et al., 2014). The composition of plastic debris in the open ocean is also suggested to be dominated by synthetic ropes and nets (Morales-Caselles et al., 2021) which represent high-risk entangling items for many species.

Coastal marine environments are often associated with higher degree of biodiversity than the open ocean (Selig et al., 2014), and subject to stressors from many anthropogenic activities (Halpern et al., 2008). In this paper, coastlines are also found to have some of the highest PAF values globally, such as the Atlantic coast off the Bahamas and several locations in Southeast Asia (Fig. 3). Moreover, marginal seas such as the Mediterranean are known plastic debris hotspots (Deudero & Alomar, 2015), and therefore stands out as an area with potential for higher impacts on marine biodiversity than other regions. In the other end of the scale, areas close to the poles are characterized by low estimated macroplastic debris concentrations, and the current potential for entanglement impacts is consequently modelled to be close to zero.

3.2.1. PAF in marine protected areas

In order to get an indication of potential impact levels within areas where anthropogenic activities are restricted (to different extents) for conservation purposes, the distribution of MPAs was considered. The potentially affected fraction of species in MPAs are relevant as these areas are often intended to safeguard vulnerable species, and when only considering the stricter categories of protection they are found to contain more at-risk biodiversity than other national waters (O'Hara et al., 2021).

The average impact level within all MPAs globally was found to be close to that of the whole ocean (global average MPA PAF: 3%, global average PAF: 4.7%). Moreover, it can be seen that many of the largest protected areas are situated in zones with low current impact levels (SI, Figure S2). This could indicate that marine fauna is in fact successfully protected within these areas, or that large MPAs may be established in distant locations where protection from many anthropogenic activities is already implicitly realized (Devillers et al., 2015). In the latter case, the MPAs do not curtail wide-reaching current pressures such as plastic pollution, which may threaten biodiversity. In addition, since the plastic debris dataset that the current results are built on was published (Eriksen et al., 2014), new measurements from the large and more remote MPAs of e.g., the South Atlantic, suggest that sea surface level concentrations have increased by 76% since 2013 (Barnes et al., 2018).

Areas where smaller MPAs overlap with high potential impact levels were identified on a regional scale, exemplified here by the Mediterranean basin (average PAF in MPAs: 13%) and the tropical northwestern Atlantic waters (average PAF in MPAs: 23%) (Fig. 4).

The PAF model suggests that species ranging across MPAs in areas with high estimated plastic debris concentrations, such as off the coast of Florida or in the eastern Mediterranean basin (Fig. 4), are not currently protected from entanglement effects of plastic debris pollution. Similarly, a recent risk assessment study considering impacts of plastic debris ingestion by marine species in the MPAs of the Mediterranean found that the present-day protection is not effective (Soto-Navarro et al., 2021). This is perhaps not unexpected, as plastic debris gets distributed according to oceanographic drivers indiscriminatory to the "borders" of protected areas, and cleanup technologies in general are still immature and not operational on a commercial scale (Bellou et al., 2021). Although plastic debris can be transported across oceans, the debris within the MPAs can still be of local origin, coming from adjacent coastal cities or maritime activities. In the Florida Keys, the same level of marine debris was found inside and outside the local MPAs, and most of it can be linked to commercial and recreational fishing activities in the area (Renchen et al., 2021). To reduce local pressures, these authors suggest tackling the littering problem at the source by increased local and regional management, in addition to providing relevant education. However, the potential impact hotspots identified in this study are found to span across national jurisdictions (Fig. 3), and so does wide-ranging marine fauna. As such, alleviating plastic pollution impacts in general and entanglement effects in particular, is likely to also require more directed multi-national governance measures aiming at stopping the inflow of mismanaged non-degradable plastic than what is currently in

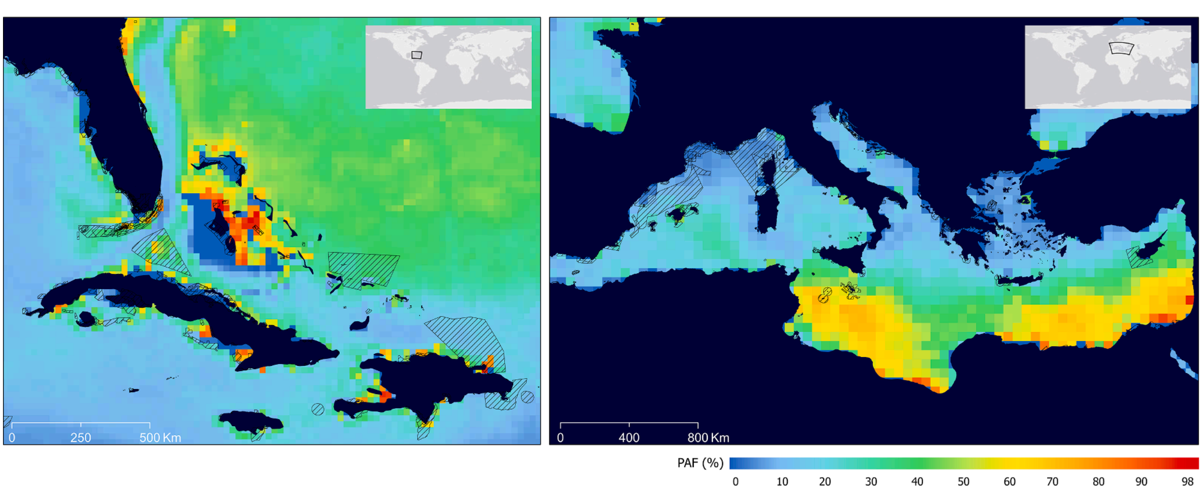

Fig. 4. Potentially affected fraction (PAF; see color gradient) of species and Marine Protected Areas (hatched areas) in tropical northwestern Atlantic (left) and the Mediterranean (right).

practice.

3.3. Sensitivity analysis

The entanglement rates derived from literature as well as the spatially explicit estimates of macroplastic debris concentrations constitute the foundation of the model and changes in these parameters should thus largely influence the output. Owing to the linear approach of the concentration–response calculations, any fixed percentage increase in the plastic debris concentration exposure or entanglement rate leads to a proportional change in the EC5. However, the extent of the potential exposure areas determines the concentrations used to derive the EC5 and will not change linearly as the concentrations are spatially heterogenous. Increasing the size of potential exposure areas led to larger changes in the EC5 for some species than others; the Antarctic fur seal, which comes out as the most sensitive species in this model, experienced the largest increase in EC5 (219%), but the resulting EC5 value remained the lowest in the subset. As such, the species' position on the curve was not influenced (SI, Figure S3).

For most of the other species, the change in EC5 value following increase in area was smaller (median absolute value of change: 9.5%) and negative (SI, Table S3). As such, the increased exposure areas did not cause a significant change in the EC5 values (Wilcoxon signed rank test, p-value = 0.15), and species that shifted position on the curve moved by one placement maximum (SI, Figure S3). Moreover, the HC50 value changed from 7.8 to 7.5 kg/km^2. These results suggest that the model is not overly sensitive to changes in the size of exposure areas, which may be a consequence of using the average plastic debris value when calculating the respective EC5s, as this masks extreme values. This can also explain the observed decrease in the HC50 value when exposure areas were increased in size, as this further minimizes the influence of local high values. As no large shifts were observed, the model is considered robust to finer scale assumptions or bias related to the definition of exposure areas, which could arise from differences in how well monitored and understood the distribution range of the species populations are.

The reliability of the parameters estimated from the SSD model directly depends on the number and variety of species used in the model (Posthuma et al., 2001). In addition, effect thresholds such as where 50% of the species are affected (HC50) could be shifted depending on which species' sensitivities are included. As such, the inclusion of additional datapoints in the model is expected to influence the EC5s and subsequently the SSD distribution, but the impact of this would depend on the specific data and cannot be quantified for this first version of the model.

3.4. Application

The current PAF indicator gives a spatially explicit overview of what share of species that may be affected under current plastic debris concentrations, where "affected" refers to 5% of a species population becoming annually entangled. Moreover, the applied concentration estimates provide a general pattern of background levels, as inputs of plastic debris to the ocean is increasing (Lebreton et al., 2018), meaning that the PAF could subsequently increase. The current results could thus be used as a first indication to identify potential hotspots of impacts of macroplastic debris, which ultimately can aid in guiding allocation of resources to targeted plastic pollution policy responses. It should be noted that this generalized indicator with global coverage can give insight on a regional level but is not suitable for interpretations on smaller spatial scales. This would require more detailed assessments where the specific conditions and biodiversity of that given area are considered and used for validation.

A specific field of application of the SSD model is in risk assessments (RA). In the RA framework, the focus is on the exposure of a given species to the environmental concentration that leads to an effect, and the predicted unacceptable risk of this event or the concentration at which it occurs is quantified (van Straalen, 2001). This application could be interesting in the case of an acute plastic pollution event, or for a scenario analysis on the environmental impacts of increasing exposure levels. An SSD-based risk assessment could also be relevant for decision-making related to the presence of marine plastic debris and what environmental concentration of this stressor constitutes an unacceptable risk to marine ecosystems. Furthermore, marine plastic pollution and its impacts has been suggested as a planetary boundary threat (Macleod et al., 2021), but owing to knowledge gaps it remains a challenge to propose control variables for this purpose (Villarrubia-Gómez et al., 2018).

SSD-based models are also commonly applied when deriving factors for environmental impacts to be used in life cycle impact assessments (LCIA). Life Cycle Assessment (LCA) is a widely used approach to assess environmental impacts of the full life cycle of products and industrial systems (Curran, 2008), allowing for a quantification of trade-offs with the use of different materials, such as plastic. All potential impacts of mismanaged plastic waste are currently neglected in the analyses (Schweitzer et al., 2018), but some authors have attempted to include the aspect of littering (Stefanini et al., 2020; Zanghelini et al., 2020), although without accounting for the possible effects on marine animals and the ecosystem. Woods et al. (2019) begun the work with a preliminary LCA effect factor (EF) for macroplastic entanglement, but the authors explicitly noted a need for better matching of the spatial dispersion of plastic debris with relevant species distributions where entanglements have been observed. The SSD-based PAF model in this work can be applied to fill this gap and is in line with a recently developed framework for including impacts on marine ecosystems in LCA (Woods et al., 2021). More specifically, the current model can be translated into an effect factor (EF), which constitutes an essential component required in establishing a novel impact category for the area of protection (AoP) ecosystem quality. To the authors knowledge, this constitutes the first model for macroplastic effects ready to be used in impact assessments, once a separate fate model estimating the distribution patterns of a given emission of mismanaged plastic has been developed.

3.5. Future development

As a preliminary indicator of potential entanglement effects globally, the current model is linked to several sources of uncertainty concerning both the underlying data and the modelling steps and assumptions made. In the following section, some of the main limitations are discussed in relation to future options to further improving the robustness of the model and expanding its application potential.

3.5.1. Standardized entanglement rates

Owing to the differences in the input data, caution needs to be exerted when comparing the sensitivity of the species in the SSD model. A challenge associated with constructing field-based SSDs is that the model is based on limited but diverse input data, taken from directed field surveys as well as more opportunistic stranding sightings. This naturally compromises comparability, as the number of entangled individuals per a total population size is systematically going to be lower than when normalized to a sample size. However, it was considered beneficial to include both types of data as what is the "best estimate" is currently not clear, and the magnitude of the potential errors unknown. In general, a standardized approach to assessing entanglement incidents relative to a representative size is lacking (Kühn & van Franeker, 2020). It has been acknowledged that rates should ideally be corrected for observer effort, but this is rarely done (McIntosh et al 2015).

3.5.2. Increased taxonomic coverage

Most of the entanglement rates pertain to either seals or turtles, which is a consequence of what was available in the literature. This could either be due to these taxa being more susceptible to entanglement effects, or because they are simply easier to study. For example, entangled seals can be more readily observed compared to many other marine animals, as they aggregate on land-based colonies (Claro et al., 2019). Future development of the model should aim at including a broader range of taxa also susceptible to plastic debris entanglement when the appropriate data becomes available. Several observations of entanglements have been made for different fish species (Andrades et al., 2021; Nunes et al., 2018), and it is also affecting the benthic community (e.g. Edward et al., 2020). However, it remains challenging to convert these observations into rates and thus get an overview of the prevalence. To include the aforementioned taxa, a global dataset of plastic debris concentration estimates from the water column and seafloor would also be required. In fact, many of the species in the current model are also likely to encounter plastic debris in the water column or on the sea floor as they dive for food, meaning that the current concentration estimates only cover parts of their exposure.

3.5.3. Differentiations in regional biodiversity and plastic debris items

In the current model it is assumed that the subset of species is representative on a global scale. As more entanglement rates from different countries become available, the SSD-model can be subdivided into different regions only including the species relevant to the given area. This could give a more accurate representation of what fraction of species living within a region that are potentially affected as a consequence of regional marine plastic pollution. Moreover, the model can also be differentiated into plastic items based on associated risks of entanglement in order to capture how some debris items are more likely to cause an entanglement effect than others (Wilcox et al., 2016), which will also depend on the body size of the animal in relation to the size of the item. As such, a more detailed dataset of the distribution of plastic debris divided into several size or item classes would be a next step towards further improving the link between concentration and response.

3.6. Conclusions

Potential entanglement impact levels were found to be higher in oceanic gyres, along coastlines and in semi-enclosed seas of certain regions, coinciding with known plastic debris hotspots. We believe that relating estimates of plastic debris concentrations to an indicator of biodiversity impact level (the PAF), can facilitate communication to policy makers and the public regarding the damage potential of mismanaged plastic. This contributes to highlighting the global issue of entanglements and can allow for efforts to be directed to hotspot areas. The map of potential impacts in this study also supports the notion that multinational efforts are required in conserving marine biodiversity, as plastic pollution is a transboundary issue and many species range across several jurisdictions. Moreover, as the first ready to be used model translating entanglement rates and plastic debris concentrations into a global indicator of entanglement effects, this work represents an important step towards the inclusion of marine plastic debris in impact assessments. Accounting for this undesirable aspect of our current consumption and end of life treatment of plastics in impact assessments would allow for trade-offs between different material types to be assessed in a more comprehensive manner. Moreover, as the field of marine debris advances and more data becomes available, the current model can easily be updated to a more detailed version accounting for factors such as entanglement risk linked to different plastic debris items.

CRediT authorship contribution statement

Marthe A. Høiberg: Data curation, Methodology, Software, Formal analysis, Writing – original draft, Writing – review & editing. **John S. Woods:** Conceptualization, Methodology, Writing – review & editing. **Francesca Verones:** Conceptualization, Methodology, Writing – review & editing.

Declaration of Competing Interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgements

The authors kindly thank Cara McHardy for her contributions to the initial data gathering and modelling development, and Johan Berg Pettersen for providing feedback and guidance on the project.

Funding

This project has received funding from the European Research Council (ERC) under the European Union's Horizon 2020 research and innovation programme (grant agreement No 850717).

Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.ecolind.2021.108509.

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. Mar. Pollut. Bull. 81 (1), 103–115. https://doi.org/10.1016/j.marpolbul.2014.02.008.

Allyn, E.M., Scordino, J.J., 2020. Entanglement rates and haulout abundance trends of Steller (Eumetopias jubatus) and California (Zalophus californianus) sea lions on the north coast of Washington state. PLoS ONE 15 (8), e0237178. https://doi.org/10.1371/journal.pone.0237178.

Andrades, R., Trindade, P.A.A., Giarrizzo, T., 2021. A novel facet of the impact of plastic pollution on fish: Silver croaker (Plagioscion squamosissimus) suffocated by a plastic bag in the Amazon estuary, Brazil. Mar. Pollut. Bull. 166, 112197 https://doi.org/10.1016/j.marpolbul.2021.112197.

Antonelis, G., Baker, J., Johanos, T., Braun, R., Harting, A., 2006. Hawaiian monk seal (*Monachus schauinslandi*): status and conservation issues. Atoll Res. Bull. 543, 75–101.

Asmutis-Silvia, R., Barco, S., Cole, T., Henry, A., Johnson, A., Knowlton, A., Landry, S., Mattila, D., Moore, M., Robbins, J., van der Hoop, J., 2017. Rebuttal to published article "A review of ghost gear entanglement amongst marine mammals, reptiles and elasmobranchs" by M. Stelfox, J. Hudgins, and M. Sweet. Mar. Pollut. Bull. 117 (1-2), 554–555.

Barnes, D.K.A., Morley, S.A., Bell, J., Brewin, P., Brigden, K., Collins, M., Glass, T., Goodall-Copestake, W.P., Henry, L., Laptikhovsky, V., Piechaud, N., Richardson, A., Rose, P., Sands, C.J., Schofield, A., Shreeve, R., Small, A., Stamford, T., Taylor, B., 2018. Marine plastics threaten giant Atlantic Marine Protected Areas. Curr. Biol. 28 (19), R1137–R1138. https://doi.org/10.1016/j.cub.2018.08.064.

Beaumont, N.J., Aanesen, M., Austen, M.C., Börger, T., Clark, J.R., Cole, M., Hooper, T., Lindeque, P.K., Pascoe, C., Wyles, K.J., 2019. Global ecological, social and economic impacts of marine plastic. Mar. Pollut. Bull. 142, 189–195. https://doi.org/10.1016/j.marpolbul.2019.03.022.

Bellou, N., Gambardella, C., Karantzalos, K., Monteiro, J.G., Canning-Clode, J., Kemna, S., Arrieta-Giron, C.A., Lemmen, C., 2021. Global assessment of innovative solutions to tackle marine litter. Nat. Sustainability 4 (6), 516–524. https://doi.org/10.1038/s41893-021-00726-2.

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.* SINTEF report M-918. Trondheim, Norway.

Borrelle, S.B., Ringma, J., Law, K.L., Monnahan, C.C., Lebreton, L., McGivern, A., Murphy, E., Jambeck, J., Leonard, G.H., Hilleary, M.A., Eriksen, M., Possingham, H. P., De Frond, H., Gerber, L.R., Polidoro, B., Tahir, A., Bernard, M., Mallos, N., Barnes, M., Rochman, C.M., 2020. Predicted growth in plastic waste exceeds efforts to mitigate plastic pollution. Science 369 (6510), 1515–1518. https://doi.org/10.1126/science.aba3656.

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.* IUCN. https://doi.org/https://doi.org/10.2305/IUCN.CH.2019.10.en.

Byard, R.W., Machado, A., 2019. Characteristic "neck collar" injuries in Australian sea lions (Neophoca cinerea) caused by marine debris. Forensic Sci. Med. Pathol. 15 (4), 631–634. https://doi.org/10.1007/s12024-018-0060-1.

Claro, F., Fossi, M.C., Ioakeimidis, C., Baini, M., Lusher, A.L., Mc Fee, W., McIntosh, R.R., Pelamatti, T., Sorce, M., Galgani, F., Hardesty, B.D., 2019. Tools and constraints in monitoring interactions between marine litter and megafauna: Insights from case studies around the world. Mar. Pollut. Bull. 141, 147–160. https://doi.org/10.1016/j.marpolbul.2019.01.018.

Cormier, S.M., Suter II, G.W., 2013. A method for deriving water-quality benchmarks using field data. Environ. Toxicol. Chem. 32 (2), 255–262. https://doi.org/10.1002/etc.2057.

Cózar, A., Echevarría, F., González-Gordillo, J.I., Irigoien, X., Úbeda, B., Hernández-León, S., Palma, Á.T., Navarro, S., García-de-Lomas, J., Ruiz, A., Fernández-de-Puelles, M.L., Duarte, C.M., 2014. Plastic debris in the open ocean. Proc. Natl. Acad. Sci. 111 (28), 10239–10244. https://doi.org/10.1073/pnas.1314705111.

Curran, M. A. (2008). Life-Cycle Assessment. In S. E. Jørgensen & B. D. Fath (Eds.), *Encyclopedia of Ecology* (pp. 2168-2174). Academic Press. https://doi.org/https://doi.org/10.1016/B978-008045405-4.00629-7.

Day, J., Dudley, N., Hockings, M., Holmes, G., Laffoley, D., Stolton, S., Wells, S., & Wenzel, L. (2019). *Guidelines for applying the IUCN protected area management categories to marine protected areas*. https://portals.iucn.org/library/sites/library/files/documents/PAG-019-2nd%20ed.-En.pdf.

Deudero, S., Alomar, C., 2015. Mediterranean marine biodiversity under threat: Reviewing influence of marine litter on species. Mar. Pollut. Bull. 98 (1), 58–68. https://doi.org/10.1016/j.marpolbul.2015.07.012.

Devillers, R., Pressey, R.L., Grech, A., Kittinger, J.N., Edgar, G.J., Ward, T., Watson, R., 2015. Reinventing residual reserves in the sea: are we favouring ease of establishment over need for protection? Aquat. Conserv. Mar. Freshwater Ecosyst. 25 (4), 480–504. https://doi.org/10.1002/aqc.2445.

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. Springer International Publishing, pp. 41–65. https://doi.org/10.1007/978-3-319-46994-2_4.

Duncan, E.M., Botterell, Z.L.R., Broderick, A.C., Galloway, T.S., Lindeque, P.K., Nuno, A., Godley, B.J., 2017. A global review of marine turtle entanglement in anthropogenic debris: A baseline for further action. Endangered Species Research 34, 431–448. https://doi.org/10.3354/esr00865.

Edward, J.K.P., Mathews, G., Raj, K.D., Laju, R.L., Bharath, M.S., Kumar, P.D., Arasamuthu, A., Grimsditch, G., 2020. Marine debris — An emerging threat to the reef areas of Gulf of Mannar. India. *Marine Pollution Bulletin* 151, 110793. https://doi.org/10.1016/j.marpolbul.2019.110793.

Eriksen, M., Lebreton, L.C.M., Carson, H.S., Thiel, M., Moore, C.J., Borerro, J.C., Galgani, F., Ryan, P.G., Reisser, J., Dam, H.G., 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 (12), e111913. https://doi.org/10.1371/journal.pone.0111913.

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*. MSFD Technical Subgroup on Marine Litter. JRC Scientific and policy reports; EUR 26113 EN. doi:10.2788/99475.

Gall, S.C., Thompson, R.C., 2015. The impact of debris on marine life. Mar. Pollut. Bull. 92 (1), 170–179. https://doi.org/10.1016/j.marpolbul.2014.12.041.

GEBCO. (2020). *General Bathymetric Chart of the Oceans Compilation Group 2020 Grid*. https://doi.org/10.5285/a29c5465-b138-234d-e053-6c86abc040b9.

GESAMP. (2019). Guidelines for the monitoring & assessment of plastic litter in the ocean. In A. T. Peter Kershaw, 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,* (Vol. 99, pp. 130). Rep. Stud. GESAMP.

Halpern, B. S., Walbridge, S., Selkoe, K. A., Kappel, C. V., Micheli, F., D'Agrosa, C., Bruno, J. F., Casey, K. S., Ebert, C., Fox, H. E., Fujita, R., Heinemann, D., Lenihan, H. S., Madin, E. M. P., Perry, M. T., Selig, E. R., Spalding, M., Steneck, R., & Watson, R. (2008). A Global Map of Human Impact on Marine Ecosystems. *Science*, *319*(5865), 948-952. https://doi.org/doi:10.1126/science.1149345.

Hauschild, M., Huijbregts, M., 2015. Introducing Life Cycle Impact Assessment. In 1–16. https://doi.org/10.1007/978-94-017-9744-3_1.

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 (9), 1077–1080. https://doi.org/10.1016/j.marpolbul.2006.05.003.

Hoondert, R.P.J., Hilbers, J.P., Hendriks, A.J., Huijbregts, M.A.J., 2018. Deriving Field-Based Ecological Risks for Bird Species. Environ. Sci. Technol. 52 (6), 3716–3726. https://doi.org/10.1021/acs.est.7b05904.

IUCN, 2021. The IUCN Red List of Threatened Species: Spatial Data Download. International Union for Conservation of Nature and Natural Resources. Retrieved 02.02 from https://www.iucnredlist.org/resources/spatial-data-download.

IUCN & UNEP-WCMC, 2021. The World Database on Protected Areas (WDPA). Cambridge (UK): UNEP World Conservation Monitoring Centre. Retrieved 02.02 from www.protectedplanet.net.

Jensen, M.P., Limpus, C.J., Whiting, S.D., Guinea, M., Prince, R.I.T., Dethmers, K.E.M., Adnyana, I.B.W., Kennett, R., FitzSimmons, N.N., 2013. Defining olive ridley turtle Lepidochelys olivacea management units in Australia and assessing the potential impact of mortality in ghost nets. Endangered Species Research 21 (3), 241–253. https://doi.org/10.3354/esr00521.

Kühn, S., van Franeker, J.A., 2020. Quantitative overview of marine debris ingested by marine megafauna. Mar. Pollut. Bull. 151, 110858 https://doi.org/10.1016/j.marpolbul.2019.110858.

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. Springer, New York, pp. 99–139. https://doi.org/10.1007/978-1-4613-8486-1_10.

Lau, W.W.Y., Shiran, Y., Bailey, R.M., Cook, E., Stuchtey, M.R., Koskella, J., Velis, C.A., Godfrey, L., Boucher, J., Murphy, M.B., Thompson, R.C., Jankowska, E., Castillo Castillo, A., Pilditch, T.D., Dixon, B., Koerselman, L., Kosior, E., Favoino, E., Gutberlet, J., Baulch, S., Atreya, M.E., Fischer, D., He, K.K., Petit, M.M., Sumaila, U. R., Neil, E., Bernhofen, M.V., Lawrence, K., Palardy, J.E., 2020. Evaluating scenarios toward zero plastic pollution. Science 369 (6510), 1455–1461. https://doi.org/10.1126/science.aba9475.

Law, K.L., 2017. Plastics in the Marine Environment. Annual Review of Marine Science 9 (1), 205–229. https://doi.org/10.1146/annurev-marine-010816-060409.

Lebreton, L., Andrady, A., 2019. Future scenarios of global plastic waste generation and disposal. *Palgrave*. Communications 5 (1). https://doi.org/10.1057/s41599-018-0212-7.

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. Sci. Rep. 8 (1), 4666. https://doi.org/10.1038/s41598-018-22939-w.

Leung, K.M.Y., Gray, J.S., Li, W.K., Lui, G.C.S., Wang, Y., Lam, P.K.S., 2005. Deriving Sediment Quality Guidelines from Field-Based Species Sensitivity Distributions. Environ. Sci. Technol. 39 (14), 5148–5156. https://doi.org/10.1021/es050450x10.1021/es050450x.s001.

Macleod, M., Arp, H.P.H., Tekman, M.B., Jahnke, A., 2021. The global threat from plastic pollution. Science 373 (6550), 61–65. https://doi.org/10.1126/science.abg5433.

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). Mar. Mammal Sci. 35 (4), 1613–1624. https://doi.org/10.1111/mms.12590.

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. Mar. Pollut. Bull. 101 (2), 716–725. https://doi.org/10.1016/j.marpolbul.2015.10.007.

Morales-Caselles, C., Viejo, J., Martí, E., González-Fernández, D., Pragnell-Raasch, H., González-Gordillo, J.I., Montero, E., Arroyo, G.M., Hanke, G., Salvo, V.S., Basurko, O.C., Mallos, N., Lebreton, L., Echevarría, F., van Emmerik, T., Duarte, C. M., Gálvez, J.A., van Sebille, E., Galgani, F., García, C.M., Ross, P.S., Bartual, A., Ioakeimidis, C., Markalain, G., Isobe, A., Cózar, A., 2021. An inshore–offshore sorting system revealed from global classification of ocean litter. Nat. Sustainability 4 (6), 484–493. https://doi.org/10.1038/s41893-021-00720-8.

Nelms, S.E., Alfaro-Shigueto, J., Arnould, J.P.Y., Avila, I.C., Bengtson Nash, S., Campbell, E., Carter, M.I.D., Collins, T., Currey, R.J.C., Domit, C., Franco-Trecu, V., Fuentes, M., Gilman, E., Harcourt, R.G., Hines, E.M., Hoelzel, A.R., Hooker, S.K., Johnston, D.W., Kelkar, N., Kiszka, J.J., Laidre, K.L., Mangel, J.C., Marsh, H., Maxwell, S.M., Onoufriou, A.B., Palacios, D.M., Pierce, G.J., Ponnampalam, L.S., Porter, L.J., Russell, D.J.F., Stockin, K.A., Sutaria, D., Wambiji, N., Weir, C.R., Wilson, B., Godley, B.J., 2021. Marine mammal conservation: over the horizon. Endangered Species Research 44, 291–325. https://www.int-res.com/abstracts/esr/v44/p291-325/.

Nunes, J.A.C.C., Sampaio, C.L.S., Barros, F., Leduc, A.O.H.C., 2018. Plastic debris collars: An underreported stressor in tropical reef fishes. Mar. Pollut. Bull. 129 (2), 802–805. https://doi.org/10.1016/j.marpolbul.2017.10.076.

O'Hanlon, N.J., Bond, A.L., Lavers, J.L., Masden, E.A., James, N.A., 2019. Monitoring nest incorporation of anthropogenic debris by Northern Gannets across their range. Environ. Pollut. 255, 113152. https://doi.org/10.1016/j.envpol.2019.113152.

O'Hara, C.C., Frazier, M., Halpern, B.S., 2021. At-risk marine biodiversity faces extensive, expanding, and intensifying human impacts. Science 372 (6537), 84–87. https://doi.org/10.1126/science.abe6731.

Perez-Venegas, D.J., Valenzuela-Sánchez, A., Montalva, F., Pavés, H., Seguel, M., Wilcox, C., Galbán-Malagón, C., 2021. Towards understanding the effects of oceanic plastic pollution on population growth for a South American fur seal (Arctocephalus australis australis) colony in Chile. Environ. Pollut. 279, 116881 https://doi.org/10.1016/j.envpol.2021.116881.

Posthuma, L., Suter, G., & Traas, T. P. (2001). *Species sensitivity distributions in ecotoxicology* (1 ed.). Lewis Publishers. https://doi.org/10.1201/9781420032314.

Renchen, G.F., Butler, C.B., Matthews, T.R., 2021. Marine debris knows no boundaries: Characteristics of debris accumulation in marine protected areas of the Florida Keys. Mar. Pollut. Bull. 173, 112957 https://doi.org/10.1016/j.marpolbul.2021.112957.

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. Mar. Pollut. Bull. 75 (1), 259–263. https://doi.org/10.1016/j.marpolbul.2013.07.003.

Ryan, P.G., 2015. Does size and buoyancy affect the long-distance transport of floating debris? Environ. Res. Lett. 10 (8), 084019. https://doi.org/10.1088/1748-9326/10/8/084019.

Schmidt, C., Krauth, T., Wagner, S., 2017. Export of Plastic Debris by Rivers into the Sea. Environ. Sci. Technol. 51 (21), 12246–12253. https://doi.org/10.1021/acs.est.7b02368.s00110.1021/acs.est.7b02368.s002.

Schweitzer, J.-P., Petsinaris, F., & Gionfra, C. (2018). *Justifying plastic pollution: how Life Cycle Assessments are misused in food packaging policy* - A study by Zero Waste Europe and Friends of the Earth Europe for the Rethink Plastic Alliance. Available at https://friendsoftheearth.eu/publication/unwrapped-how-throwaway-plastic-is-failing-to-solve-europes-food-waste-problem-and-what-we-need-to-do-instead/.

Selig, E.R., Turner, W.R., Troëng, S., Wallace, B.P., Halpern, B.S., Kaschner, K., Lascelles, B.G., Carpenter, K.E., Mittermeier, R.A., Rotjan, R.D., 2014. Global Priorities for Marine Biodiversity Conservation. PLoS ONE 9 (1), e82898. https://doi.org/10.1371/journal.pone.0082898.

Shomura, R. S., & Yoshida, H. O. (1985, 26-29 November 1984). Proceedings of the Workshop on the Fate and Impact of Marine Debris. Honolulu, Hawaii. U.S. Dept. of Commerce.

Soto-Navarro, J., Jordá, G., Compa, M., Alomar, C., Fossi, M.C., Deudero, S., 2021. Impact of the marine litter pollution on the Mediterranean biodiversity: A risk assessment study with focus on the marine protected areas. Mar. Pollut. Bull. 165, 112169 https://doi.org/10.1016/j.marpolbul.2021.112169.

Stefanini, R., Borghesi, G., Ronzano, A., Vignali, G., 2021. Plastic or glass: a new environmental assessment with a marine litter indicator for the comparison of pasteurized milk bottles. The International Journal of Life Cycle Assessment 26 (4), 767–784. https://doi.org/10.1007/s11367-020-01804-x.

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 [Review]. *Frontiers in Marine*. Science 5 (238). https://doi.org/10.3389/fmars.2018.00238.

Thorley, J., Schwarz, C., 2018. ssdtools: An R packageto fit Species Sensitivity Distributions. Journal of Open Source Software 3 (31). https://doi.org/10.21105/joss.01082.

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. https://doi.org/10.1016/j.marenvres.2017.07.009.

van Straalen, N.M., 2001. Theory of ecological risk assessment based on species sensitivity distributions. In: Posthuma, L., Suter, G., Traas, T.P. (Eds.), Species sensitivity distributions in ecotoxicology, 1 ed. Lewis Publishers, pp. 37–48. https://doi.org/10.1201/9781420032314.

Villarrubia-Gómez, P., Cornell, S.E., Fabres, J., 2018. Marine plastic pollution as a planetary boundary threat – The drifting piece in the sustainability puzzle. Marine Policy 96, 213–220. https://doi.org/10.1016/j.marpol.2017.11.035.

Wallace, B.P., DiMatteo, A.D., Hurley, B.J., Finkbeiner, E.M., Bolten, A.B., Chaloupka, M. Y., Hutchinson, B.J., Abreu-Grobois, F.A., Amorocho, D., Bjorndal, K.A., Bourjea, J., Bowen, B.W., Dueñas, R.B., Casale, P., Choudhury, B.C., Costa, A., Dutton, P.H., Fallabrino, A., Girard, A., Girondot, M., Godfrey, M.H., Hamann, M., López-Mendilaharsu, M., Marcovaldi, M.A., Mortimer, J.A., Musick, J.A., Nel, R., Pilcher, N.J., Seminoff, J.A., Troëng, S., Witherington, B., Mast, R.B., Bograd, S.J., 2010. Regional Management Units for Marine Turtles: A Novel Framework for Prioritizing Conservation and Research across Multiple Scales. PLoS ONE 5 (12), e15465. https://doi.org/10.1371/journal.pone.0015465.

Waluda, C.M., Staniland, I.J., 2013. Entanglement of Antarctic fur seals at Bird Island. South Georgia. *Mar Pollut Bull* 74 (1), 244–252. https://doi.org/10.1016/j.marpolbul.2013.06.050.

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*. MSFD GES Technical Group on Marine Litter. JRC Technical report; EUR 28317 EN. doi:10.2788/690366.

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. https://doi.org/10.1016/j.marpol.2015.10.014.

Woods, J.S., Verones, F., Jolliet, O., Vázquez-Rowe, I., Boulay, A.-M., 2021. A framework for the assessment of marine litter impacts in life cycle impact assessment. Ecol. Ind. 129, 107918 https://doi.org/10.1016/j.ecolind.2021.107918.

Zanghelini, G.M., Cherubini, E., Dias, R., Kabe, Y.H.O., Delgado, J.J.S., 2020. Comparative life cycle assessment of drinking straws in Brazil. J. Cleaner Prod. 276, 123070 https://doi.org/10.1016/j.jclepro.2020.123070.