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Risks of floating microplastic in the global ocean

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ABSTRACT

Despite the ubiquitous and persistent presence of microplastic (MP) in marine ecosystems, knowledge of its potential harmful ecological effects is low. In this work, we assessed the risk of floating MP (1 μm −5 mm) to marine ecosystems by comparing ambient concentrations in the global ocean with available ecotoxicity data. The integration of twenty-three species-specific effect threshold concentration data in a species sensitivity distribution yielded a median unacceptable level of $1.21 * 10^5$ MP m^{-3} (95% CI: $7.99 * 10^3$–$1.49 * 10^6$ MP m^{-3}). We found that in 2010 for 0.17% of the surface layer (0–5 m) of the global ocean a threatening risk would occur. By 2050 and 2100, this fraction increases to 0.52% and 1.62%, respectively, according to the worst-case predicted future plastic discharge into the ocean. Our results reveal a spatial and multidecadal variability of MP-related risk at the global ocean surface. For example, we have identified the Mediterranean Sea and the Yellow Sea as hotspots of marine microplastic risks already now and even more pronounced in future decades.

1. Introduction

Microplastic (MP) comprises small pieces of plastic debris (1 μm–5 mm), and is found at increasing concentrations in marine ecosystems all over the globe (Fahrenkamp-Uppenbrink, 2018; Wilcox et al., 2020). Microplastic particles are known to wash ashore (Cozar et al., 2014; Turra et al., 2015), sink to the seafloor (Martin et al., 2017; Van Cauwenberghe et al., 2013), be ingested by organisms (Cole et al., 2013; Thompson et al., 2004; van Franeker et al., 2011), be packed in sea ice (Bergmann et al., 2019), or be both airborne (Bouwmeester et al., 2015; Panko et al., 2013) and/or waterborne (van Sebille et al., 2015). Efforts have been undertaken in the past two decades to quantify *in situ* concentrations of MP, and in sites such as harbours (Noren, 2007), lagoons (Vianello et al., 2013), and straits (Desforges et al., 2014), the MP concentration is relatively high (Lusher, 2015), leading to concerns on their ecotoxicological risk to local populations and communities (Besseling et al., 2019; Everaert et al., 2018).

Despite the awareness of the MP ubiquity and persistence in the environment, the impact on marine organisms exposed to plastic particles remains poorly understood. The currently available ecotoxicity data are often difficult to use interoperably as: (1) MP particles have a wide size range (Burns and Boxall, 2018; Hartmann et al., 2017); (2) plastic particles have variable shapes which are often not reported (Koelmans et al., 2017); (3) plastic particles include a wide range of polymers with distinct physic-chemical properties and composition (Koelmans et al., 2017); (4) concentrations of plastic particles are reported in different units (Koelmans et al., 2017); and (5) plastic particles can either be primary produced virgin particles or secondary products resulting from natural weathering, which impacts the surface shape and properties, and impacts the leaching rate of contaminants (Koelmans et al., 2017). Because of the uncertainty associated with the ecotoxicological results, the current reports on risk assessment of MP in the aquatic environment (Adam et al., 2019; Besseling et al., 2019; Burns and Boxall, 2018; Everaert et al., 2018; Zhang et al., 2019) should be considered preliminary (ECHA, 2019).

The current studies on the risk assessment of MP in the aquatic

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environment suggest that the *in situ* concentrations are on average several orders of magnitude lower than the concentrations where effects are expected to occur (Science Advice for Policy by European Academies, 2019). However, Everaert et al. (2018) and Besseling et al. (2019) pointed out that in areas where the current concentration of MP exceeds unacceptable levels (MP concentrations above which adverse ecological effects are likely to occur), such as in harbours, narrow straits, and in coastal regions, local organisms and populations are likely already at risk. There is a need to identify hotspots of risk to prioritise mitigation measures, as MP concentrations are expected to increase in the future (Geyer et al., 2017; Science Advice for Policy by European Academies, 2019), which will lead to increased areas where such risks will occur (Everaert et al., 2018). In order to obtain a realistic risk assessment, the European Chemicals Agency (ECHA) suggests integrating quantifiable uncertainty and variability into the risk assessment paradigm through probability distributions (ECHA, 2012). A probabilistic risk assessment combines a probability distribution for *in situ* concentrations with a probability distribution for ecotoxicological data (Solomon et al., 2000). As such, a probabilistic risk assessment makes maximal use of the variability in the data and results for the studied environmental compartment. This approach was recently used by Adam et al. (2019) to assess the risk of MP in freshwater and resulted in a sounder risk assessment than if performed using single values or fitting mathematical models to multiple data. A probabilistic approach allows inclusion of all data available at one point in time and gives an overall picture of the situation as it is known, and eases the risk quantification.

Here, we examine the risk of floating MP in the ocean surface by integrating environmental MP concentrations with ecotoxicity data according to the risk assessment paradigm (Koelmans et al., 2017), i.e. the standard way to assess risks of chemicals under the REACH legislation (EU, 2016). We first quantified unacceptable levels of MP concentrations based on ecotoxicity data available in scientific literature. In parallel, we quantified past, current and future environmental concentrations of MP based on distribution data provided by van Sebille et al. (2015). To draw conclusions about the past, current and future risk of MP at the ocean surface [< 5 m depth], we compared *in situ* MP concentrations with the corresponding unacceptable levels using a probabilistic approach.

2. Materials and methods

We quantified the global risk of MP in the marine environment. From the perspective of environmental sciences risk has beeen defined as “the probability of an adverse effect on the environment occurring as a result of a given exposure to a substance” (van Leeuwen and Vermeire, 2007). Under the European REACH legislation, a risk assessment for environmental pollutants is composed of an effect assessment and an exposure assessment. The environmental risk assessment (ERA) paradigm is a validated approach that has been applied in a range of MP studies. For instance, Koelmans et al. (2017) provided a template, employing adverse outcome pathways and tiered hazard assessment strategies to systematize and quantify the microplastic issues at hand, and this resulted in well-accepted applications in e.g. Everaert et al. (2018), Besseling et al. (2019), Burns and Boxall (2018), Zhang et al. (2019), Adam et al. (2019) and Skåre et al. (2019). In our probabilistic approach, we compared the probability density function (PDF) of *in situ* concentrations with the PDF of ecotoxicity data as in Schad and Schulz (2011). By doing so, we integrated the natural variability of *in situ* concentrations and the uncertainty due to the relative low amount of available ecotoxicity data.

2.1. Environmental MP concentrations

The global MP concentrations provided by van Sebille et al. (2015) are available in a 1° by 1° grid for distinct plastic sizes (0.15 mm–200 mm, of which > 90% in the 0.33 mm–200 mm range) for the year 2014. The size distribution of a large part of the data in van Sebille et al. (2015) is documented in the publication of Cozar et al. (2014). The size frequency distribution follows an exponential behaviour (i.e. Fig. 3 in Cozar et al. (2014)), in line with later meta-analysis based findings of Kooi and Koelmans (2019). The present work focusses on waterborne MPs in the surface of the ocean (<5 m). A depth integration of particle numbers performed by Kooi et al. (2016) showed that including the surface (0 m) to 5 m depth increases the microplastic numbers significantly compared to surface data only (Kooi et al., 2016). Reisser et al. (2015) found in a multi-level trawl in the North Atlantic gyre that plastic concentrations drop exponentially with water depth; hence, deeper sampling is likely to be superfluous when the main interest is in the surface layer.

In order to obtain yearly estimates (between 1950 and 2100) of MP concentrations in the upper ocean surface layer (0 m–5 m) in the same spatial grid for the entire MP size range (i.e. size of 1 μm–5 mm) three calculations are needed: (1) depth integration; (2) particle size integration; and (3) temporal extrapolation. We converted the spatial distribution data provided by van Sebille et al. (2015) from their original unit to MP m^{-3} in seawater. The data of van Sebille et al. (2015) are expressed as the amount of particles per size unit (km^{-2}) and we recalculated these towards amount of particles per volume (m^{-3}) of seawater based on the average sampling depth of a Manta trawl (i.e. 0.25 m; van Sebille et al., 2015). This conversion resulted in concentrations of microplastics in the 0 m–0.25 m layer. Next, we used the vertical distribution of buoyant plastic in the ocean (Kooi et al., 2016) to extrapolate the top 0.25 m concentrations to 0 m - 5 m depth MP concentrations. As numerical plastic *in situ* concentrations in the upper 5 m mixed zone are typically underestimated by a factor 2.74 (95% CI: 1.04–30.0) when using a Manta trawl (Kooi et al., 2016), we multiplied the MP concentrations from the 0 m - 0.25 m layer by a factor 2.74 in each 1° by 1° grid by a factor 2.74. In a next step, the size frequency distribution of Kooi and Koelmans (2019) (Eq. (1)) was used to change the original size range of the ambient concentrations (van Sebille et al., 2015) (0.15 mm–200 mm) to the MP size range (1 μm–5 mm; Data S1).

$$y = bx^{-\propto} \quad (1)$$

In Eq. (1), x is the particle size (μm), $y$is the abundance of particles, b is a coefficient, and $\propto$ is an exponent with a mean value (±Standard Deviation, SD) of 1.6 ± 0.5 (Kooi and Koelmans, 2019). In Data S1, we report the change of the relative abundance for different MP sizes with $\propto = 1.1$ and $\propto = 2.1$. The MP concentrations of van Sebille et al. (2015) represented the year 2014, but we aimed for MP distribution maps for each year between 1950 and 2100. Hence, we made a temporal extrapolation of the MP distribution map. To achieve this, we calculated a plastic production polynomial function based on global plastic production data between 1950 and 2016 (PlasticsEurope, 2016) using the method of Koelmans et al. (2016). We found a good model fit ($R^2 = 0.99$) between observed and predicted plastic production data, and details about the polynomic function are available in Supplementary Materials (Data S2). From the polynomial function, we inferred global plastic production estimates between 1950 and 2013, and between 2015 and 2100. To do so, we assumed a business as usual scenario until 2100, meaning that future global plastic production will follow the same polynomial trend as the one inferred between

1950 and 2014 (Data S2). As the spatial patterns of the van Sebille distribution model (van Sebille et al., 2015) will not change with lower or higher amounts of MP, we back calculated (1950–2014) and predicted (2015–2100) the spatial coverage of the MP concentrations in a global 1° by 1° grid. The three previous steps resulted in yearly (from 1950 to 2100) estimates of the concentration of plastic particles (MP m^{-3}; 1 μm–5 mm) in a 1° by 1° grid at global scale. Based on these ambient MP concentrations, we calculated a PDF for four years between 1950 and 2100 (i.e. 1970, 2010, 2050, and 2100). The global yearly estimates of MP pollution are available in an interactive interface: http://rshiny.lifewatch.be/ng-ocean-plastic-challenge.More than 90% of the observations used in van Sebille et al. (2015) are based on Manta trawls taken in the top 0.25 m surface layer of the ocean, hence by deduction mainly floating and buoyant polymers such as polyethylene (PE), polypropylene (PP), and polystyrene (PS) are included (Reisser et al., 2015; Erni-Cassola et al., 2019).

2.2. Ecotoxicity data

The risk assessment paradigm for microplastic is a tool aimed at characterizing and quantifying the potential risks of microplastic particles and provides indispensable insight in terms of addressing questions of risk. A robust risk assessment is obtained if high quality scientific data are integrated (Gouin et al., 2019). We strictly curated the data that were integrated in the risk assessment based on stringent criteria to guarantee input of high quality data (Table S1). We queried ecotoxicity effect data from scientific literature indexed in the Web of Science (WoS) until August 1st, 2019 and we aimed to obtain marine ecotoxicity data relevant to be compared with our environmental MP concentrations (Table 1). The focus of our study was on direct particle effects of MP and the role of MP as a vector of organic pollutants with associated potential adverse impact was not addressed in our analysis, as it is considered of minor importance (Koelmans et al., 2016; Science Advice for Policy by European Academies, 2019). The initial search in the WoS on “marine” AND “microplastic*” resulted in 1604 publications. This selection was reduced to 211 publications after querying for “impact*” OR “toxic*” OR “effect*” OR “exposure*” OR “hazard*” OR “affect*” in the article title or abstract. Based on an abstract cross-read of these publications, studies dealing with freshwater or terrestrial ecosystems, only focussing on ingestion of plastics by organisms, and/or only reporting about *in situ* sampling were discarded. From the remaining 111 publications we retained those that studied effects on individual level endpoints that have clear links to population dynamics i.e. growth, mortality, reproduction, survival, fecundity, hatching success, weight, length, and offspring. To ensure a good match between *in situ* concentrations and ecotoxicity data, studies focussing on plastic particles smaller than 1 μm were omitted (15 publications in total). Of the remaining 51 studies, only those using commonly used floating polymers (Erni-Cassola et al., 2019) i.e. PE, PS, and PP, were used. Inspired by the criteria in Hartmann et al. (2017), Moermond et al. (2016) and Connors et al. (2017), we evaluated the adequacy of the remaining publications to be used in an ecotoxicological risk assessment. Twelve quality control criteria were used (see Table S1 for more information).

In total, 23 ecotoxicity studies were selected after this review process. Concentrations of the dilution series in effect studies reported in mass-based units were converted to number of particles per volumetric unit (no. particles L^{-1}) based on the mean radius, the polymer density, and assuming a spherical shape, as per Everaert et al. (2018). Chronic no observed effect concentrations (NOEC) and lowest observed effect concentrations (LOEC) were

Table 1
Data on microplastic (MP) effects in marine species used in the effect assessment. Types of plastic used: polystyrene (PS), polyethylene (PE), and polypropylene (PP).

Test species	Effect threshold concentration (no. particles L^{-1})	Endpoint	Polymer type	MP size (μm)	Reference
Phylum Cnidaria					
Acropora spp.	1900	Coral bleaching and necrosis	PE	37–163	Reichert et al. (2018)
Phylum Mollusca					
Mytilus edulis	110,000	Metabolic rate	PS	10–90	Cauwenberghe et al. (2015)
Ostrea edulis	63,240	Abundance and biomass	PE	0.48–316	Green (2016)
Pinctada margaritifera	160	Growth & reproduction	PS	6–10	Gardon et al. (2018)
Magallana gigas	100,000	Growth	PS	1	Cole and Galloway (2015)
Mytilus galloprovincialis	230,000	Growth	PE	1–50	Détrée and Gallardo-Escárate (2018)
Crepidula onyx	10,000	Growth	PS	2	Lo and Chan (2018)
Phylum Arthropoda					
Calanus helgolandicus	37,500	Feeding rate	PS	20	Cole et al. (2015)
Palaemon pugio	50,000	Survival	PE	30–165	Gray and Weinstein (2017)
Euphausia superba	116,000	Survival and weight	PE	27–32	Dawson et al. (2018)
Centropages typicus	2,000,000		PS	7.3	Cole et al. (2013)
Artemia parthenogenetica	10,000,000	Growth	PS	10	Wang et al. (2019)
Phylum Echinodermata					
Tripneustes gratilla	100,000	Growth	PE	25–32	Kaposi et al. (2014)
Paracentrotus lividus	500,000	Larval development	PS	6	Martínez-Gómez et al. (2017)
Phylum Chordata					
Acanthurus triostegus	5000	Survival	PS	91	Jacob et al. (2019)
Oryzias melastigma	50,000	Survival	PS	10	Cong et al. (2019)
Ciona robusta	113,700	Survival	PS	10	Messinetti et al. (2017)
Ciona intestinalis	45,500,000,000	Growth	PS	1	Messinetti et al. (2019)
Sebastes schlegelii	500,000	Growth	PS	15	Yin et al. (2018)
Phylum Ochrophyta					
Skeletonema costatum	2,240,000	Growth	PS	74	Zhu et al. (2019)
Chaetoceros neogracile	900,000,000	Growth	PS	2	Long et al. (2017)
Phylum Rotifera					
Brachionus plicatilis	62,700,000	Mortality	PE	1–4	Beiras et al. (2018)
Phylum Haptophyta					
Isochrysis galbana	1,410,000,000	Growth	PE	3.29	Garrido et al. (2019)

inferred according to European Union (EU) legislation (EU, 2016). If several chronic NOEC or LOEC values for different toxicological endpoints were available for a single species, the lowest value was used. LOEC values were converted to NOEC values by dividing them by two (Organisation for Economic Co-operation and Development, 1995). The sensitivity of organisms is expected to follow a Gaussian curve, and a species sensitivity distribution (SSD) reflects the observation that interspecies difference in sensitivity to a stressor resemble a bell-shape distribution on a log-normal axis (Posthuma et al., 2019). SSDs are commonly used in risk assessment to describe the sensitivity of different species to a substance, to identify the most sensitive species and to derive environmental quality criteria. In the present research, the SSD of the NOEC values was estimated using a lognormal model as described by Aldenberg and Jaworska (2000) and implemented as by Szöcs (2015), using the *fitdistrplus* R package (Delignette-Muller and Dutang, 2015) in the free statistical software R (R Core Team, 2018). The mean hazardous concentration for 5% of the species (HC5) and a bootstrapped confidence interval around the HC5 were derived using 1000 random parameterisations of the distribution. As stipulated in the EU legislation, the regulatory safe concentration, known as predicted no effect concentration (PNEC), was calculated from the HC5 using an assessment factor (AF) of five (EU, 2016). The Monte Carlo simulation based on 1000 iterations of the lognormal model served as a basis of the empirical PDFs of the ecotoxicity data, such as the PNEC. Due to limited amount of information on the size class of MP, we did not infer size-specific PNECs, and the risk estimates do not account for potential ingestion. As the amount of high-quality ecotoxicity data for the pelagic compartment was limited, we complemented the pelagic ecotoxicity data with data of other marine taxa. In ERA, pragmatic pooling of available data is often done if data are limited (Skåre et al., 2019), and based on the log-normal distribution of species' sensitivity. The combination of taxa from different habitats and ecosystem types is strictly for calculation purposes, and does not imply that they are supposed to share the same habitat (Besseling et al., 2019).

Risk assessments often use risk characterization ratios (RCRs), whereby a risk is characterized as the ratio of actual or predicted exposures to the maximum acceptable concentration of a given chemical or particle in a given environment. An RCR exceeding 1 is usually interpreted by policymakers as an unacceptable situation that warrants further study and/or risk mitigation measures. We considered marine communities to be at risk in an ecosystem if the safe concentration, i.e. unacceptable level, was exceeded.

2.3. Probabilistic risk assessment

The probabilistic risk distribution (PRD) was calculated as the logarithmic ($\log_{10}$) difference between the distribution of MP concentrations and the distribution of effect concentrations following Aldenberg et al. (2001) and Verdonck et al. (2003) (Eqs. (2) and (3)). Environmental risks are to be expected if $\log_{10}$ PRD >0.

$$\mathrm{Log}_{10}\mathrm{ECD} - \log_{10}\mathrm{SSD} \sim \mathrm{Normal}(\mu_{ECD} - \mu_{SSD},\ \sqrt{\sigma^2_{ECD} + \sigma^2_{SSD}}) \quad (2)$$

$$\mathrm{Log}_{10}\mathrm{PRD} = \log_{10}\mathrm{ECD} - \log_{10}\mathrm{SSD} \quad (3)$$

This analysis was performed at a global level, but in case particular regions showed results deviating from the global mean, they were assessed in greater detail. In particular, we expected that the Mediterranean Sea would deviate from the global mean, based on Cózar et al. (2015) and Compa et al. (2019).

2.4. Mapping of risks

We quantified the risk of MP for each year between 1950 and 2100, and we determined when the first risk occurred somewhere in the ocean. We mapped the quantified risks for four distinct years (i.e. 1970, 2010, 2050, and 2100), and for creating the global maps we used a best case (safe concentration is at its maximum value), median case (safe concentration is at median value), and worst case (safe concentration is at minimum value) scenario. Global risks of MP were displayed in a four-panel plot, in which each panel corresponded to a specific year (i.e. 1970, 2010, 2050, and 2100). Cell specific (i.e. 1° by 1°) risk estimates were categorized in ten classes and transformed to the conformal Spilhaus oceanic map projection (Fig. S1) using NASA GISS's global map projector *G. Projector*. To ease the interpretation, we mapped the cell specific risk estimates using a Robinson Projection (Robinson, 1974) (Fig. S2). In addition, a 3D visualization of the data was generated in Esri ArcScene. The risk estimates were represented in 3D as elevation values with a vertical exaggeration of 0.1. To quantify the ocean surface at risk, we used the *geosphere* package (Karney, 2013) in R, taking into account the curvature of the Earth. By doing so, each 1° by 1° grid cell in the *in situ* data of van Sebille et al. (2015) was attributed a certain ocean surface. To quantify the actual ocean surface at risk due to MP, we summed the surface area of the individual grid cells that are at risk. Finally, the surface area expected to be at risk in a certain year was expressed relatively to the total ocean surface area. Expressing the risks of MP in terms of ocean surface can partly give results deviating from the probabilistic approach, as a 1° by 1° grid cell has different surfaces at different latitudes.

3. Results

3.1. Exposure assessment based on ambient microplastic concentrations

We have estimated that the highest microplastic (MP, 1 μm–5 mm) concentrations in the ocean surface layer (0–5 m) in 2014 were located in the Eastern part of the Mediterranean Sea and accounted for over 40,000 MP m^{-3} (10th quantile, Q10 = 126 MP m^{-3} - 90th quantile, Q90 = 13,446 MP m^{-3}) and of over 50,000 MP m^{-3} in the Yellow Sea [close to the Chinese mainland] (Q10 = 255 MP m^{-3}, Q90 = 5626 MP m^{-3}). In 2014, 80% of the MP concentrations in the surface layer were between 0.001 MP m^{-3} (Q10) and 67.9 MP m^{-3} (Q90). Based on the plastic production polynomial function (Data S2), and under a business as usual scenario, we calculated that by 2100 the annual plastic production will be of 1800 MT (Data S2), which represents a 400% increase compared to the current annual plastic production of 335 MT (PlasticsEurope, 2016). Based on the size frequency distribution of Kooi and Koelmans (2019) and its power exponent (α) of 1.6 ± 0.5, we calculated that for each plastic particles with a size between 330 μm and 20 mm, a mean of 48 MPs between 1 μm and 330 μm are present (95% CI = 2.65–1.03 * 10^3; Data S1).

3.2. Effect assessment based on ecotoxicity data

Effect data for 23 different species from eight phyla were included in our assessment: *Arthropoda, Chordata, Cnidaria, Echinodermata, Haptophyta, Mollusca, Ochrophyta*, and *Rotifera* (Table 1). It is noteworthy that the toxicity data of the phyla Chordata, Mollusca and Arthropoda span a relatively wide range between the minimum and maximum species-specific effect threshold concentrations (Fig. 1 and Table 1). Amongst others, for phyla such as Ochrophyta, Rotifera, Haptophyta continued efforts are needed to perform ecotoxicity tests at lower concentrations (i.e.

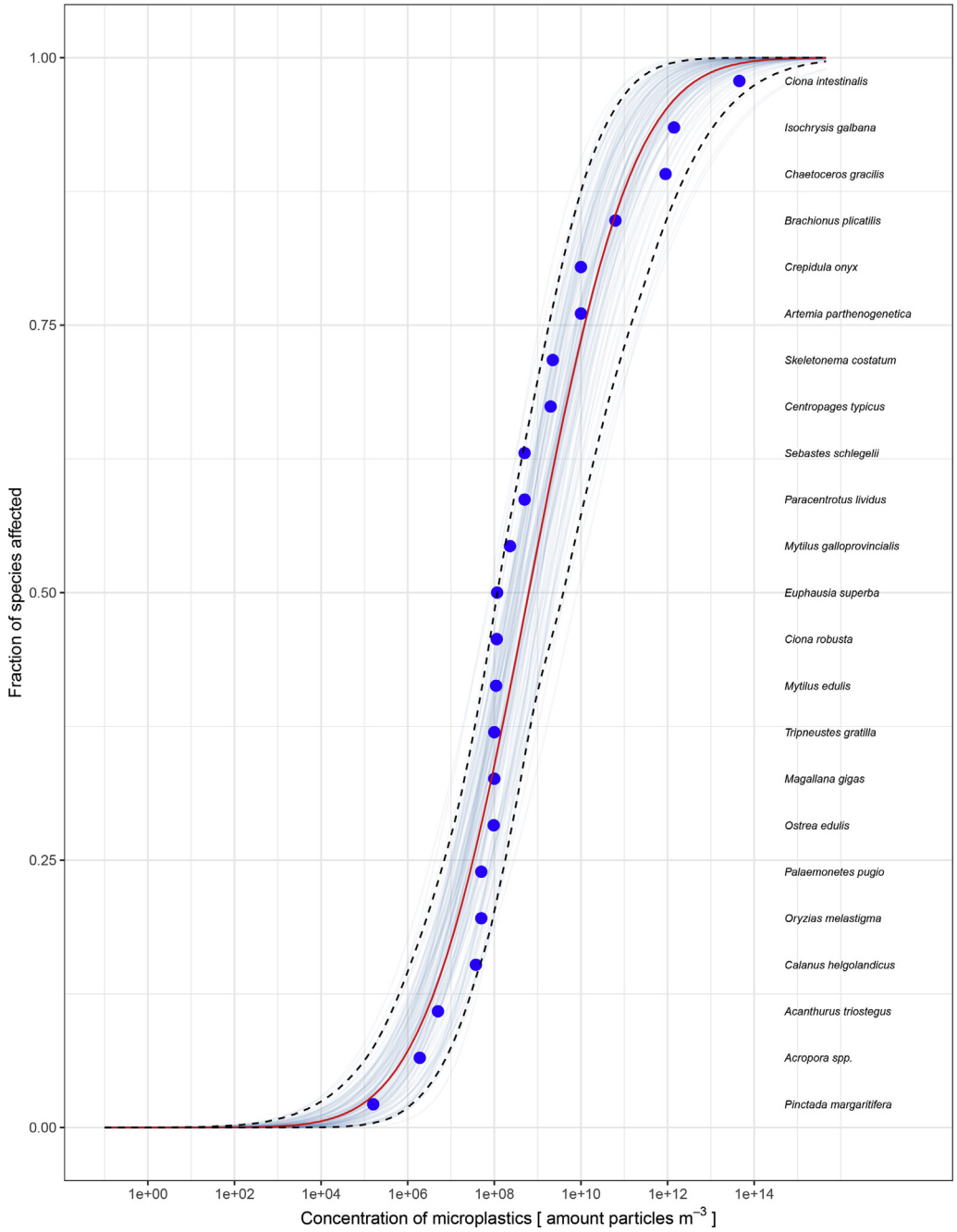

Fig. 1. Species sensitivity distribution (SSD) for buoyant microplastic (MP m^{-3}) based on effect data listed in Table 1. A log-normal distribution is fitted, and the labels indicate the species that was exposed to microplastic. Detailed information on the effect data from literature, i.e. marine species exposed to microplastic, the size of the particles, the type of plastics used, and the endpoints used to assess the potential effects are available in Table 1 and supplementary materials. Blue dots are NOECs of each species. The actual species sensitivity distribution is depicted in red and is surrounded by a confidence interval (black dotted lines) derived using 1000 random parameter iterations (grey lines) of the lognormal distribution. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

concentrations that are not orders of magnitude higher than ambient concentrations). Out of the 23 studies that have been included eight have been performed with PE, and fifteen studies were performed with PS (Table 1). Popular MP sizes that have been tested are between 1 μm and 100 μm (Table 1). The resulting median unacceptable level (Predicted No-Effect Concentration, PNEC) was 1.21 * 10^5 MP m^{-3} (95% CI: 7.99 * 10^3 MP m^{-3} – 1.49 * 10^6 MP m^{-3}; Fig. 1).

3.3. Probabilistic risk assessment

We estimated that MP pollution did not pose an immediate risk to marine ecosystems in the 1970s (Fig. 2), but the MP concentrations approach the unacceptable level with increasing plastic production. For example, the probability that marine organisms are affected by MP pollution in 2010 was estimated to be at a probability of 0.51%, of an overlap between the *in situ* MP concentrations and the ecotoxicity probability distribution (Table 2). By 2050 and

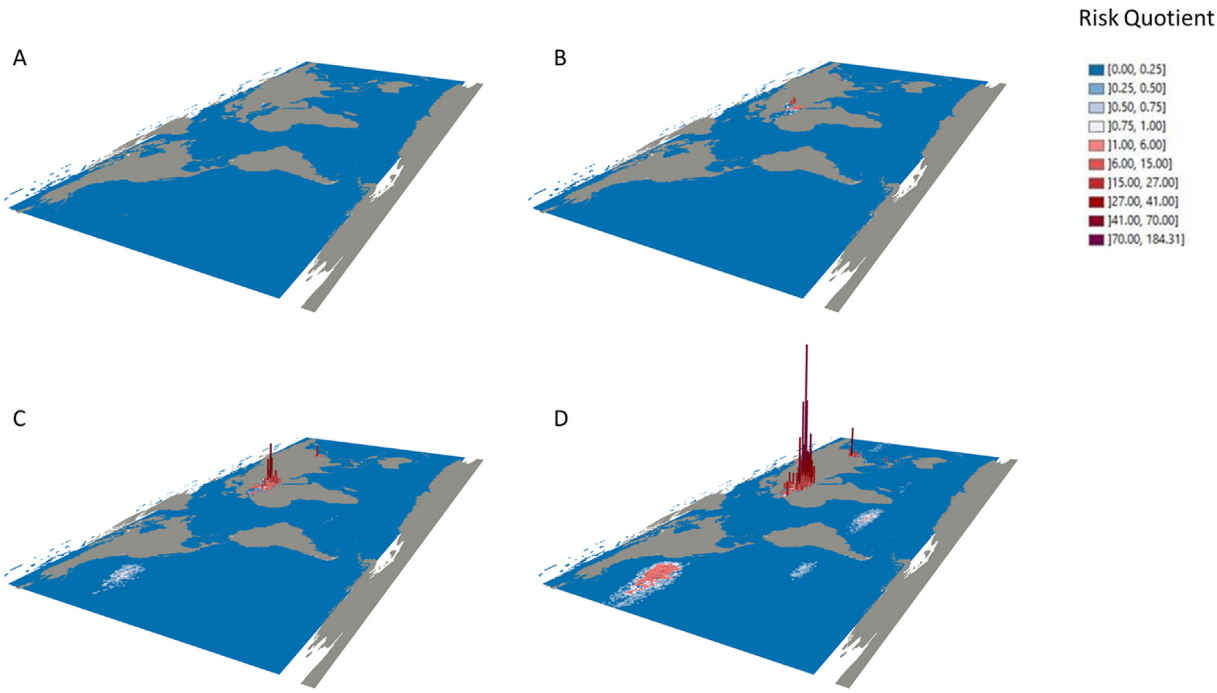

Fig. 2. Global risks of microplastic pollution based on worst case scenario (unacceptable level (PNEC) = 7.99 *10^3 MP m^{-3}) displayed in a four-panel plot, in which each panel corresponded to a specific year: 1970 (A), 2010 (B), 2050 (C), and 2100 (D). To do so, cell specific (1° by 1°) risk estimates were calculated and a 3D visualization of the data was generated. The risk estimates were represented in 3D as elevation values. As long is the risk quotient remains lower than the value of 1 (bluish tones), policy makers consider no risk due to MPs. In case that the risk quotient exceeds the value of 1 (reddish tones), there is a risk. Variations on this figure are available in supportive information Fig. S1 and S2.

Table 2
Risk expressed in a probabilistic manner, in the global ocean surface layer (0–5 m), the Mediterranean Sea, and the Yellow Sea.

Region and year	Probabilistic risk assessment (% overlap between *in situ* concentration distribution and ecotoxicity distribution)	Area affected (% of the ocean surface where *in situ* concentrations are expected to outrange unacceptable levels, i.e. PNEC)		
		Best case scenario (Unacceptable level = 1.49 * 10^6 MP m^{-3})	Median case scenario (Unacceptable level = 1.21 * 10^5 MP m^{-3})	Worst case scenario (Unacceptable level = 7.99 * 10^3 MP m^{-3})
Global				
1970	0.06	0.00	0.00	0.00
2010	0.51	0.00	0.00	0.17
2050	1.16	0.00	0.03	0.52
2100	2.06	0.00	0.21	1.62
Mediterranean Sea				
1970	0.87	0.00	0.00	0.00
2010	5.37	0.00	0.00	15.9
2050	10.2	0.00	1.27	44.6
2100	15.7	0.00	19.9	68.7
Yellow Sea				
1970	2.86	0.00	0.00	0.00
2010	12.4	0.00	0.00	5.38
2050	20.0	0.00	1.77	27.1
2100	27.1	0.00	1.77	53.9

2100, the fraction of ocean surface where it is probable that the MP concentrations exceed the unacceptable level of MP concentration increased to 1.16% and 2.06%, respectively. These figures are in line with the small percentage (0.12%) of the probabilistic distribution currently exceeding the unacceptable levels in freshwater ecosystems (Adam et al., 2019). We found that currently 0.17% of the global ocean surface layer is impacted by MP pollution and that this fraction will increase to 1.62% by 2100 under worst case scenario (Table 2).

We found strong indications that organisms in parts of the Mediterranean Sea and the Yellow Sea (Fig. 2) are currently at risk. In 2010, organisms in 15.9% and 5.38% of the surface area of the Mediterranean Sea and the Yellow Sea, respectively, were exposed to MP concentrations exceeding the unacceptable level of 7.99 * 10^3 MP m^{-3} (Table 2). By 2100, we expect that 68.7% and 53.9% of the Mediterranean Sea and the Yellow Sea will have unfavourable conditions for marine life due to MP pollution, under a worst case and in a business as usual scenarios, respectively. The North Pacific Ocean gyre was identified as an area that contains substantial amounts of MP, at a mean ± SD of 6408 ± 6020 MP m^{-3} (Q10 = 787 MP m^{-3} – Q90 = 15,324 MP m^{-3}), enough to put the corresponding ecosystems at risk (Fig. 2). Although marine organisms will not be at risk by 2100 in the gyre of the South Atlantic Ocean, MP concentrations approaching the unacceptable levels are

probable, at a mean ± SD of 2513 +- 2157 MP m^{-3} (Q10 = 336 MP m^{-3} – Q90 = 5355 MP m^{-3}) (Fig. 2). For the Arctic Sea, as well as the Southern Ocean, our analysis indicates neither current nor future risks (Fig. 2).

4. Discussion

Our results showed substantial spatial differences in the risk of MP in the ocean surface layer (0–5 m depth), and we identified the Mediterranean and Yellow seas as current hotspots of risks. The Mediterranean Sea had been reported to contain high loads of plastics compared to other regional seas (Compa et al., 2019; Cózar et al., 2015). The high human pressure, together with the hydrodynamics of this semi-enclosed basin, makes it a plastic accumulation area with no outflow possibilities of marine litter except to greater depths. Our results show a low risk of MP in polar regions, which should be considered with precaution, as they can be limited by the low amount of MP data from these areas. Air (Bergmann et al., 2019) and sea ice (Peeken et al., 2018) have recently been identified as possible transport vectors for MP in and to the Arctic, and the current MP load in polar areas may be larger than expected. Despite the lack of direct sources of plastics in polar regions, these ecosystems are likely to be exposed to MP, and the MP ecotoxicological risk should be re-evaluated in the future.

We calculated an unacceptable level, i.e. a Predicted No-Effect Concentration (PNEC), of 1.21 * 10^5 MP m^{-3} which falls within the confidence interval of a previously assessed PNEC (Everaert et al., 2018) of 70–2.8 * 10^6 MP m^{-3} in marine ecosystems. In assessments combining both freshwater and marine species, Besseling et al. (2019) found a PNEC between 1.01 * 10^5 MP m^{-3} - 1.02 * 10^7 MP m^{-3} for aquatic ecosystems, and Burns and Boxall (2018) reported a PNEC of 1.28 * 10^7 MP m^{-3} (PNEC = HC5/5; reported HC5 = 6.4 * 10^7 MP m^{-3}). In freshwater, Zhang et al. (2019) reported a PNEC for surface water of 4920 MP m^{-3}, and Adam et al. (2019) found a mean safe concentration of 7.4 * 10^5 MP m^{-3}. In Everaert et al., 2018, the acceptable level for marine benthic ecosystems is only based on a single effect point. As more effect data becomes available, the two orders of magnitude for confidence intervals of the current risk estimates are expected to narrow down and we expect the reliability of future estimates to improve. The adverse effects to be expected in case that ambient concentrations exceed unacceptable levels are not defined exactly, but should considered as adverse effects on the ecosystem function and ecosystem structure. At the extreme, exceeding of unacceptable levels can lead to major adverse outcomes such as: 1) population-level alteration (including demographic bottlenecks and stressor-induced selection); 2) changes in genetic diversity; and 3) changes in evolutionary trajectories (Chapman et al., 2016).

To date, we do not know how MP exposure interacts with other global change stressors, and so future ecotoxicological work should urgently explore the contribution of MP to the multiple stressor conundrum in marine ecosystems. We estimated that the ecotoxicological risk of marine communities exposed to MP is low. However, an equal amount of MP in an oligotrophic open ocean with low biomass and low biodiversity may result in lower ecosystem impact than in more diverse and biomass-rich upwelling zones or reef structures, subject to other stressors (e.g. ocean acidification, climate change, emerging and other contaminants, and fisheries) (Rockström et al., 2009). The risk of MP needs to be addressed within a multiple-stressors scenario, so governmental bodies can be confident that research and remediation measures are prioritised, and so policymakers can objectively rank information and implement measures to tackle realistic environmental risks.

In the present research, we applied the environmental risk assessment paradigm to quantify the risk of organisms to MP exposure (Koelmans et al., 2017; Leeuwen, 2007), according to the European risk assessment guidelines. Even thought we did not integrate biomass or biodiversity distribution data, our results clearly show is that harmful effects of MP cannot be excluded in marine ecosystems. Using well-established risk assessment methodologies, we can conclude that MP pollution causes an environmental risk in about 1% in the global ocean surface layer (0–5 m). Global mapping of the MP risks is instrumental to identify marine regions that need increased attention for mitigation measures. We suggest that ecosystems that are enclosed through either currents (gyres) and/or topography, are more at risk, such as the Mediterranean and Yellow Seas. We strongly recommend that environmental managers consider the predicted rise of microplastic concentrations, as an increasing number of ecosystems will experience higher chance of exposure to unacceptable levels of MP in the near future.

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.

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Appendix A. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.envpol.2020.115499.

Credit author statement

Gert Everaert - Conceptualization, Investigation, Methodology, Formal analysis, Writing - original draft; Maarten De Rijcke – Conceptualization, Writing - original draft, Writing - review & editing; Britt Lonneville – Visualization, Software; Colin Janssen – Conceptualization, Methodology; Thomas Backhaus – Methodology, Writing - review & editing; Jan Mees – Conceptualization, Funding acquisition; Erik van Sebille – Methodology, Data curation, Writing - review & editing; Albert A. Koelmans – Methodology, Writing - review & editing; Ana I. Catarino - Methodology, Writing - review & editing; Michiel Vandegehuchte –Writing - review & editing, Supervision

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