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Risk assessment of microplastics in the ocean: Modelling approach and first conclusions☆

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ABSTRACT

We performed an environmental risk assessment for microplastics (<5 mm) in the marine environment by estimating the order of magnitude of the past, present and future concentrations based on global plastic production data. In 2100, from 9.6 to 48.8 particles m^{-3} are predicted to float around in the ocean, which is a 50-fold increase compared to the present-day concentrations. From a meta-analysis with effect data available in literature, we derived a safe concentration of 6650 buoyant particles m^{-3} below which adverse effects are not likely to occur. Our risk assessment (excluding the potential role of microplastics as chemical vectors) suggests that on average, no direct effects of free-floating microplastics in the marine environment are to be expected up to the year 2100. Yet, even today, the safe concentration can be exceeded in sites that are heavily polluted with buoyant microplastics. In the marine benthic compartment between 32 and 144 particles kg^{-1} dry sediment are predicted to be present in the beach deposition zone. Despite the scarcity of effect data, we expect adverse ecological effects along the coast as of the second half of the 21st century. From then ambient concentrations will start to outrange the safe concentration of sedimented microplastics (i.e. 540 particles kg^{-1} sediment). Additional ecotoxicological research in which marine species are chronically exposed to realistic environmental microplastic concentration series are urgently needed to verify our findings.

1. Introduction

Given the rise of microplastic related studies (Barboza and Gimenez, 2015; Kramm et al., 2018), our knowledge of microplastic pollution in the marine environment has significantly increased over the past decades, but the environmental or ecological (including human) risks of microplastics in marine environments have, to date, not been addressed and quantified. Numerous monitoring campaigns of different marine systems have revealed that microplastics are ubiquitous and can be present at very high concentrations (Van Sebille et al., 2015; Auta et al., 2017; Chen et al., 2018; Cole et al., 2011; Eriksen et al., 2014). In comparison to these monitoring efforts that result in exposure concentrations, effect studies aiming to quantify the (adverse) effects of microplastics are much rarer and may suffer from a number of shortcomings: (1) the data collected does not always allow to evaluate the potential effects of microplastics (i.e. environmental risk cannot be assessed solely based on monitoring or ingestion data), (2) most of the laboratory studies are poorly standardised, and (3) few of these studies are performed with the aim to establish a concentration-response relationship (needed to derive effect thresholds). Additionally, (adverse) effects of microplastic exposure have mostly been investigated by administering unrealistically high (usually single-dose) concentrations of microplastics to test organisms (Van Cauwenberghe, 2015; Koelmans et al., 2017). Most effect-oriented laboratory studies were aimed at demonstrating the ingestion or uptake of microplastics by marine organisms (Lusher et al., 2015). From these observations it is then often inferred that microplastic do cause adverse ecological effects and, hence, are an imminent threat to the marine environment. However, these type of statements and the classification of microplastics as contaminants of concern have often been made without adequate, scientific

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confirmation (Koelmans et al., 2017). While this approach may provide some initial, useful insights into potential effects, testing at more relevant (i.e. ambient) concentrations and using different test concentrations to establish a concentration-response relationship is needed to provide more appropriate information to assess the present and future risk of microplastics for marine ecosystems (Van Cauwenberghe, 2015; Koelmans et al., 2017). The existing frameworks for assessing environmental risks of pollutants (e.g. ecological risk assessment), which are used in regulatory contexts worldwide, are yet to be applied to marine microplastics. Such a generic ecological risk assessment is composed of an exposure assessment, an effect assessment, and a risk characterisation (Koelmans et al., 2017; EU, 2006), and objectively determines the risk of a contaminant to (marine) ecosystems. Until present such an approach has not yet been applied to microplastics, and so we are not able to univocally demonstrate whether microplastic contamination poses a risk to the marine environment. Therefore, in the present research, we have performed an ecological risk assessment of marine microplastic pollution based on data available in literature. To do so, we calculated the current exposure of marine systems to microplastics and simulated how this exposure will evolve in the coming century. An effect assessment was used to quantify the safe environmental concentration of microplastics i.e. the predicted concentration below which adverse effects are not expected to occur. By combining the results of both the exposure and the effect assessment (i.e. in the risk characterization step), we are able to assess whether microplastics constitute a risk for marine biota and their ecosystems. Hence, the risk assessment presented here will, for the first time (Koelmans et al., 2017), combine literature data on exposure to and potential effects of microplastics and answer, albeit tentative, the question: do microplastics pose a risk to marine pelagic and marine benthic ecosystems?

2. Materials and methods

Two potentially adverse effects are commonly discerned when the impacts of microplastics on the marine environment are discussed, being (1) the direct effects of microplastic exposure, and (2) the indirect effects associated with the chemicals present in and on microplastics. In the present research, the primary focus is on the risk assessment of direct effects of microplastic exposure. Regarding the role of microplastics as a vector of organic pollutants, we refer to recent studies by (Bakir et al., 2017), Ziccardi et al. (2016) (Ziccardi et al., 2016) and Koelmans et al. (2016) (Koelmans et al., 2016), but note that there is some discussion about the potential toxicological harm (Lohmann, 2017; Hartmann et al., 2017).

2.1. Exposure assessment

Environmental concentrations of microplastics

Based on historical annual total plastic production figures of 1950–2016 ($PL_{mass,t}$ in Eq. (1)), and a projected annual global growth of 4.5% in plastics production between 2017 and 2100 ($1.045 * PL_{mass,t-1}$ in Eq. (2)), (Plastics Europe, 2016) the past and future concentrations of microplastic were calculated. To do so, some assumptions were made based on literature data. The world synthetic fibre production data were not included in our exposure assessment. First, we assumed that 1.75% to 4.62% of the mass of the total yearly plastic production becomes marine litter (f_{ML}) (Jambeck et al., 2015). Secondly, we assumed that 94% of the marine litter that enters the sea ends up on the seabed ($f_{seabed} = 94\%$), whereas five percent washes ashore ($f_{ashore} = 5\%$) and the remaining part becomes free floating microplastics ($f_{float} = 1\%$) (Sherrington, 2016). So, by multiplying f_{ML} with these fractions (i.e. f_{seabed}, f_{ashore}, and f_{float}) the fraction of the total mass of plastic that enters a specific compartment can be calculated. For example, between 0.0175% (i.e. 1% of 1.75%) to 0.0462% (i.e. 1% of 4.62%) of all plastic that is produced on a mass basis will become free-floating marine microplastics. By doing so, we inherently assume that all plastic that ends up in the ocean will eventually become microplastics. The rates of weight loss due to solar radiation and oxygenation (Andrady Bergmanet al., 2015) of frequently used plastic polymers were assumed to vary between 0.65% and 5% per year (Artham et al., 2009; Sudhakar et al., 2007), and are represented by *Weight loss*. As such, the mass of microplastic in a certain environmental compartment (i.e. free-floating marine pelagic (f_{float}), washed ashore (f_{ashore}) and seabed (f_{seabed})) from a specific year t ($MP_{mass,t}$ in 10^6 tonnes) in the past (from 1950 to 2016) was calculated using the following equation:

$$MP_{mass,\,t} = \left[MP_{mass,\,t-1} + \left(PL_{mass,t} * f_{ML} * f_{(float,\ \ ashore,\ \ seabed)}\right)\right] * (1 - Weight\ loss) \quad (1)$$

The future (from 2017 to 2100) microplastic mass in these compartments in a specific year t ($MP_{mass,t}$ in 10^6 tonnes) was calculated using the following equation:

$$MP_{mass,\,t} = \left[MP_{mass,\,t-1} + \left(1.045 * PL_{mass,t-1} * f_{ML} * f_{(float,\ \ ashore,\ \ seabed)}\right) \times\right] * (1 - Weight\ loss) \quad (2)$$

A worst case scenario (minimum annual weight loss of 0.65% and maximum annual marine litter production rate of 4.62%) and a best case scenario (maximum weight loss of 5% and minimum marine litter production rate of 1.75%) were developed for all three environmental compartments.

From the mass-based amount of microplastics present in the marine environment (i.e. Eq. (1) and Eq. (2)), we calculated the number of microplastic particles on a weight basis (i.e. number of particles per kg sediment) and on a volumetric basis (i.e. number of particles per m^3 seawater). For free-floating microplastics, the amount of particles was expressed as number of particles per m^3 of seawater (Predicted Environmental Concentration i.e. $PEC_{t,\ float}$ in Eq. (3)). To do so, the total number of buoyant microplastic particles ($n_{particles,t}$) present in the marine environment in a specific year t was divided by the total volume of seawater ($Volume_{ocean}$) in the ocean. In this context, the surface of the ocean was estimated to be 3.62×10^8 km (Kramm et al., 2018), (Eakins and Sharman, 2010) and buoyant microplastics are assumed to be concentrated in the upper 5 m layer of the water column (Kooi et al., 2016; Reisser et al., 2015). Hence, the $Volume_{ocean}$ in which the microplastics are present is about 1.81×10^{15} m^3.

$$PEC_{t,\ float} = \frac{n_{particles,\ t}}{Volume_{ocean}} \quad (3)$$

The total number of buoyant microplastic particles ($n_{particles,t}$) was quantified by dividing the buoyant microplastic mass ($MP_{mass,t}$ from Eq. (1) and Eq. (2)) in a specific year t by the average mass of one microplastic particle (Eq. (4)).

$$n_{particles,\ t} = \frac{MP_{mass,\ t}}{Mass_{particle}} \quad (4)$$

The average mass of one microplastic particle ($Mass_{particle}$) was quantified by multiplying the average density of those plastic types

that are most commonly found free-floating (i.e. polypropylene (PP) and polyethylene (PE) with a mean density of 0.925 g cm^{-3}; 1930 kg m^{-3} (Daly et al., 1966). Hence, the $PEC_{t,\ ashore}$ can be calculated as:

$$PEC_{t,\ ashore} = \frac{n_{particles,\ t}}{Length_{coast} * Width_{deposition\ zone} * Depth_{deposition\ zone,\ \ ashore} * Density_{sediment}} = \frac{n_{particles,\ t}}{Mass_{sediment}} \quad (6)$$

$Density_{particle}$) (Van Cauwenberghe, 2015), and the estimated weighted mean volume per particle ($Volume_{particle}$) as described in Eq. (5).

$$Mass_{particle} = Density_{particle} * Volume_{particle} \quad (5)$$

For estimating the mean weighted volume per particle, we assumed three size classes of microplastics based on Desforges et al., 2014) (Desforges et al., 2014): i.e. lower range small microplastics (LR-SMPs; < 500 μm), upper range small microplastics (UR-SMPs; 0.5–1 mm), and large microplastics (LMPs, 1–5 mm). These three size classes represent different fractions of microplastics in the natural environment: 10% of the number of microplastic particles are LMPs, 15% of the number of microplastic particles are UR-SMPs, and the remaining 75% of the number of microplastic particles are LR-SMPs (Desforges et al., 2014; Nor and Obbard, 2014; Song et al., 2014). Assuming normal size distributions within each class, we set the average particle size per class at 250 μm (LR-SMPs), 750 μm (UR-SMPs), and 3000 μm (LMPs). These diameters were then converted to mean volumes per particle assuming that the fragments and pellets (which contribute for 94.4% to the microplastics mass) (Kooi et al., 2016) are spheres. While calculating we assumed that the size class fractions (i.e. 10% for LMPs, 15% for UR-SMPs, and 75% for LR-SMPs) remained constant over time, and that the weight loss rates were independent from the size of the particles. Finally, we converted the total volumetric amount of microplastics to total particle numbers per size class using a weighted mean volume per microplastic, and the natural relative abundance of each size class from literature.

For microplastics that wash ashore, the amount of particles was expressed as number of particles per kg dry sediment (i.e. $PEC_{t,\ ashore}$ in Eq. (6)). To do so, the total number of marine beached microplastic particles ($n_{particles,t}$) in a specific year t was divided by the estimated sediment mass in which they are expected to end up ($Mass_{sediment}$). The latter was calculated based on the total global length of coastline ($Length_{coast}$; 1.63×10^6 km) (WRI, 2000), and assuming that: (1) an average beach deposition zone has a width of 50 m ($Width_{deposition\ \ zone}$) (Chubarenko and Stepanova, 2017; Enríquez et al., 2017; Madzena and Lasiak, 1997; Turra et al., 2014) (2) all microplastics are present in the top 0.4 m of the coastal sediment ($Depth_{deposition\ zone,\ ashore}$) (Moreira et al., 2016), and (3) the mean density of dry beach sediment ($Density_{sediment}$) is about

The total number of microplastic that wash ashore ($n_{particles,t}$) is quantified based on Eq. (4). The average mass of one microplastic particle that washes ashore ($Mass_{particle}$) is quantified by multiplying the average density of those plastic types that are commonly found ashore (i.e. polypropylene (PP), polyethylene (PE), polyvinylchloride (PVC), polystyrene (PS) and nylon with a mean density of 1.078 g cm^{-3}; $Density_{particle}$) (Van Cauwenberghe, 2015), and the estimated weighted mean volume per particle ($Volume_{particle}$) as described in Eq. (5).

For microplastics that end up on the seabed, the amount of particles was expressed as number of particles per kg dry sediment (i.e. $PEC_{t,\ seabed}$ in Eq. (7)). To do so, the total number of marine seabed microplastic particles ($n_{particles,t}$) in a specific year t was divided by the estimated sediment mass in which they are expected to end up ($Mass_{sediment}$). The latter was calculated based on the surface of the global ocean ($Surface_{ocean}$; 3.62×10^8 km^2) (Eakins and Sharman, 2010), and assuming that: (1) all microplastics are present in the top 2.5 cm of the seabed ($Depth_{deposition\ zone,\ seabed}$) (Martin et al., 2017) and (2) the mean density of seabed sediment ($Density_{sediment}$) is about 1440 kg per m^3 (Daly et al., 1966). Hence, the $PEC_{t,\ seabed}$ can be calculated as:

$$PEC_{t,\ \ seabed} = \frac{n_{particles,\ t}}{Surface_{ocean} * Depth_{deposition\ zone,\ \ \ seabed} * Density_{sediment}} = \frac{n_{particles,\ t}}{Mass_{sediment}} \quad (7)$$

Similar as for the microplastics that wash ashore, also for the microplastics that sink to the seafloor (i.e. PVC, PS, and nylon; mean density of 1.180 g cm^{-3}) the total number of microplastic ($n_{particles,t}$) is quantified based on Eq. (4). The average mass of one microplastic particle ($Mass_{particle}$) that ends up on the seabed is calculated in the same way as for a particle that washes ashore (cfr. previous paragraph).

Environmental concentrations of microplastics in marine bivalves

In 2014, the amount of microplastics in blue mussels (*Mytilus edulis*) from Belgian marine waters was 0.36 particles g ww^{-1} ($MP_{body\ \ burden,2014}$) (Van Cauwenberghe et al., 2015). Future microplastic concentrations in year t in *M. edulis* ($MP_{body\ \ burden,t}$) were quantified by multiplying the above-mentioned body burden ($MP_{body\ \ burden,2014}$) with an accumulation factor. This accumulation factor was the ratio between the order of magnitude estimation of the amount of free-floating microplastics in the year of interest (PEC_t) and the estimated amount of free-floating microplastics in 2014 (PEC_{2014}; Eq. (8)).

$$MP_{body\ burden,\ t} = MP_{body\ burden,\ 2014} * \frac{PEC_t}{PEC_{2014}} \quad (8)$$

Two scenarios were considered with regard to the indirect ingestion of microplastics by humans through the consumption of *M. edulis*; one for European top consumers (26.3 kg y^{-1}), and one for Europeans that have a low per capita consumption of shellfish (4.3 kg y^{-1}) (EFSA, 2011).

2.2. Effect assessment

For the effect assessment, we scanned the available scientific literature for effect data that expose marine organisms to microplastics. Chronic no observed effect concentrations (NOEC) and chronic lowest observed effect concentrations (LOEC) were inferred according to European Union (EU) legislation (EU, 2006). 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 2 (OECD, 1995). The species sensitivity distribution (SSD) of the NOEC values was developed using a lognormal model as described by Aldenberg and Jaworska (2000) (Aldenberg and Jaworska, 2000) and implemented by Szöcs (2015) (Szöcs, 2015) using the *fitdistrplus* package in the free statistical software R (R Development Core Team, 2015). The mean HC_5 (hazardous concentration for 5% of the species) and a confidence interval around the HC_5 were derived using 1000 random parameter iterations of the distribution. As stipulated in the EU legislation, the safe concentration, also known as predicted no effect concentration (PNEC), was calculated from the HC_5 using an assessment factor (AF) of 1–5 (EU, 2006). Such AFs are often applied to effect data to yield a dose or concentration to which humans or organisms may be exposed that is expected to be safe.

2.3. Risk characterisation

The final step in the environmental risk assessment is the risk characterisation (Van Cauwenberghe, 2015; Koelmans et al., 2017). To do so, a risk characterisation ratio (RCR; Eq. (9)) is calculated as the ratio of the PEC and the PNEC.

$$RCR = \frac{PEC}{PNEC} \quad (9)$$

When this RCR is < 1 no immediate risk for the environment is discerned, as environmental concentrations are lower than the concentration below which adverse effects will most likely not occur (i.e. the PNEC). Increasing environmental concentrations will subsequently result in the increase of the RCR. A RCR >1 indicates that environmental concentrations are exceeding the safe concentration defined by the PNEC, and it is concluded that a risk to the environment cannot be excluded.

3. Results and discussion

3.1. Environmental concentrations of microplastics

Microplastics are ubiquitously present in all compartments of the marine environment. Assuming minimal input of plastics into the marine environment and maximal weight loss, we found a total global mass of 4.9×10^5 tonnes of floating microplastics in 2010 (Fig. S1). This theoretical estimate is in line with predictions of Eriksen et al. (2014) (i.e. 2.7×10^5 tons) who based their estimates on *in situ* observations. We predict that by 2100 the mass of floating microplastics) will increase to 2.5×10^7 to 1.3×10^8 tonnes (best case and worst case scenario, respectively), i.e. a 50-fold increase in the total microplastic mass between 2010 and 2100 (Fig. S1). Converting these mass-based data to concentrations, we retrospectively quantified microplastics concentrations in 2010 ($PEC_{2010,\ float}$) of 0.2–0.9 particles m^{-3} (Fig. 1A). By 2100, we predict an increase to concentrations ranging from 9.6 to 48.8 particles m^{-3} for the best and worst case scenario, respectively (Fig. 1A). Comparison between our predicted environmental concentrations and *in situ* observations leads to three important observations. First, the concentrations of buoyant microplastics predicted by our model follow the increasing trend of increasing *in situ* microplastic concentrations (Fig. 1A). However, the observed concentration of free-floating microplastic highly depends on the sampling site (Fig. 1A). For example, 0.27 particles m^{-3} were found in the Western English Channel (Cole et al., 2014), while in NE Pacific seawater samples concentrations from 8 to 9200 particles m^{-3} were found (Desforges et al., 2014). Despite the simplification of our theoretical model that all plastic is considered microplastic one year after its release in the marine environment, a second important observation is that our predictions are within the range of *in situ* microplastic concentrations for each of the environmental compartments (Fig. 1A). As a third point, the predicted environmental concentrations do not span the entire concentration range of the ambient marine conditions, especially in the pelagic compartment, due to the high spatial variability of the observed *in situ* concentrations of free floating microplastics. One of the main reasons for this discrepancy is that in the theoretical calculations of the present research, we covered the entire size range of microplastics (from <1 μm to 5 mm) as we start from the entire global plastic production and do not have a lower limit to the definition of microplastics. However, when performing *in situ* monitoring only particles larger than the mesh size are retained. Hence, the smallest microplastics are missed as they slip through the mesh of the nets and are thus not counted. Those small particles have a large contribution in the total mass (Enders et al., 2015) and in the total number of microplastics particles expressed per volumetric unit (Kang et al., 2015). It was found that the mean concentrations when using a mesh size of 50 μm was two orders of magnitude greater than when using a sampling net of 330 μm (Kang et al., 2015). (Norén, 2007) even collected 100,000 times more microplastics with an 80-μm mesh net than with a 450-μm mesh net. The phenomenon of 'missing' microplastic debris has already been described earlier and is of fundamental scientific and societal importance for understanding the microplastic cycling in the marine environment (Cozar et al., 2017). It has been hypothesized that fast nano-fragmentation of microplastics into particles of microns or smaller as well as their transfer to the ocean interior including sea ice, food webs and ballasting processes, such as biofouling are amongst the most important loss mechanisms (Cozar et al., 2017; Obbard et al., 2014; Woodall et al., 2014; Bergmann et al., 2017). As these potentially important sinks of microplastics have not been taken into account in our theoretical model, the latter may be a second reason for over-predicting of certain environmental concentrations by our exposure model (Fig. 1A). Amongst others, biased project-based *in situ* sampling towards areas with high plastic loads (i.e. ocean gyres, harbours and beaches) is a plausible reason for the under-prediction of certain concentrations by our exposure model.

Predicted environmental concentrations of microplastics in oceanic sediments (PEC_{seabed}; Fig. 1B) and microplastics that wash ashore (PEC_{ashore}; Fig. 1C) are expected to increase in future years. The increasing trend found in our exposure assessment is in line with the work of Claessens et al. (2011) We retrospectively found an increase of microplastic concentrations of a factor 2 to 2.5 (depending on the scenario) between 1993 and 2008 (Fig. 1C). In

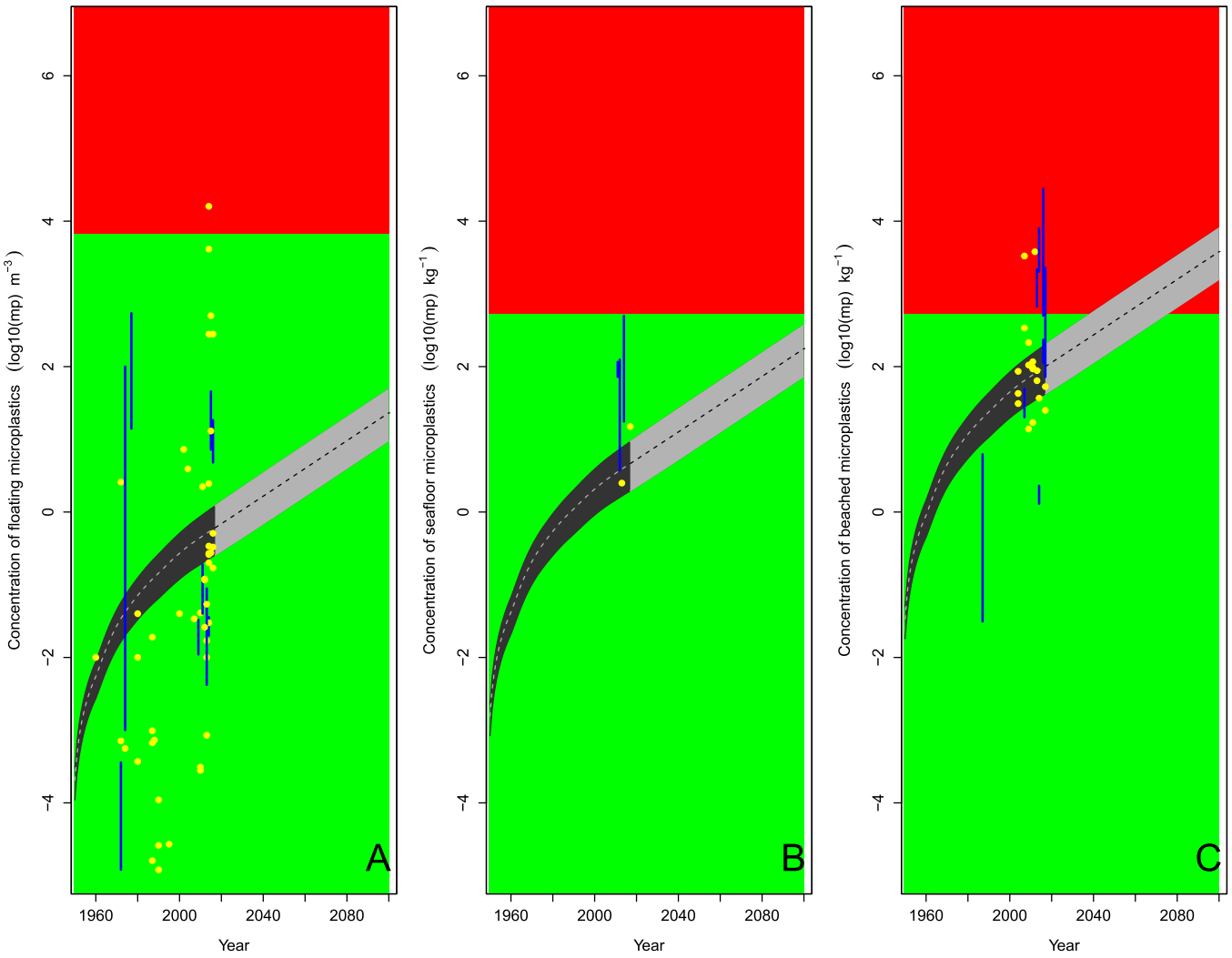

Fig. 1. Past, present and future projections of the concentration of global marine free-floating microplastics (panel A), the concentrations of microplastics that end up on the seabed (panel B), and the concentration of microplastics that wash ashore (panel C) in the marine environment. Historic retrospective microplastic abundances (pre-2016) are represented by the black polygon, while future predicted abundances (2017–2100) are depicted in grey. The dotted line represents the average predicted concentrations and is surrounded the best (lower) and worst (upper) case scenario. Yellow dots are actual *in situ* observations as reported in scientific literature (see List S1 for all references used). If a concentration range was reported in a certain study, a blue line was drawn between the minimum and maximum reported concentration. Measured and predicted environmental concentrations at which no adverse ecological effects of microplastics are to be expected are plotted against a green background. A red background indicates that the safe concentration as calculated in the present study was exceeded, hence adverse ecological effects are likely to occur at these sites. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

the same period Claessens et al. (2011) reported an increase by a factor 2.5 to 3 along Belgian beaches. To date, microplastic concentrations are about twenty fold higher in the intertidal sediments (32–144 particles kg^{-1}) compared to deep sea sediments (1.5–6.7 particles kg^{-1}) in 2010. This trend persists with time and by 2100, our predictions yield minimal concentrations of 1580 particles kg^{-1} and maximum concentrations of 8050 particles kg^{-1} on beaches, while deep sea sediment concentrations range from 73 to 373 particles kg^{-1} in the best and worst case scenario, respectively. For sedimented microplastics our theoretical model has a better overlap with *in situ* monitoring data (Fig. 1B and C) than in the pelagic environment (Fig. 1A). The latter relates to the sampling methodology that is often used in marine sediments. Indeed, quantifying the amount of microplastics in sediments is often based on a density separation that extract virtually all plastic particles from the sediment (Claessens et al., 2013). This technique is more efficient than towing a sampling net that will inherently loose those particles that are smaller than the mesh size (Kang et al., 2015). As the amount of microplastics particles increases exponentially with decreasing particle size (Song et al., 2014), a high amount of particles gets lost during sampling in the marine pelagic compartment.

Increasing environmental concentrations of microplastics enhance the probability of organisms encountering and interacting with microplastics. This will most likely lead to more ingestion of microplastics in the future (Figs. S2 and S3). It is well established that, due to their small sizes, microplastics are available for ingestion by a wide array of marine biota. While microplastic ingestion was already demonstrated in laboratory experiments dating back to the early onset of microplastic research (Thompson et al., 2004; Browne et al., 2008), it was not until recently that the accumulation in organisms collected from natural systems, i.e. exposed to environmental plastic concentrations, was demonstrated (Murray and Cowie, 2011; de Witte et al., 2014; Mathalon and Hill, 2014; Van Cauwenberghe and Janssen, 2014). In addition to the body

burden of microplastics in mussels, we also quantified the human exposure to microplastics through their consumption. As the soft body of bivalves is eaten completely and may contain high levels of microplastics, bivalves could potentially constitute an important source of microplastics to humans. Van Cauwenberghe et al. (2015) (Van Cauwenberghe et al., 2015) reported an average concentration of microplastics in mussels of 0.36 particles g ww^{-1}. Given a consumption of 2.4 kg mussels y^{-1}, the latter values suggest that current-day consumers may ingest up to 864 microplastic particles y^{-1}. For an average European (EFSA, 2011), being key consumers of shellfish, this number may vary between 1550 and 9474 microplastics y^{-1}, depending on their (low-high) consumption pattern (Fig. S2). Based on the predicted (relative) increase of PEC_t and the accumulation factor, we expect the body burden of mussels to increase to 15.8 microplastic particles g ww^{-1} by 2100 (Fig. S3). The increased accumulation in bivalves by 2100 will result in an increased intake by humans: the consumption of Europeans will increase to 6.6×10^4 particles y^{-1} for minor and up to 4.4×10^5 particles y^{-1} for top consumers (Fig. S2). To date, the long-term impact of microplastics on human health remains largely unknown as most studies have been limited to the ingestion by and impact on marine life (Kontrick, 2018). As calculated in the present research and already indicated earlier in literature, e.g. (Van Cauwenberghe et al., 2015) it is clear that humans are exposed to microplastics via consumption of seafood, but so far the consequences and potential risks have not yet been quantified. We extrapolated the average microplastic body burden of few individual mussels (n = 36) to all mussels that are consumed in one portion. Recent work of Hermsen et al. (2017) in fish however (i.e. 400 individual fish of four North Sea species) showed that some individual fish ingested microplastics, but most of them (i.e. > 99%) did not. As such, our results originating from the linear extrapolation of ingestion rates should be interpreted carefully. In terms of microplastics being a potential vector of toxic chemicals there is some discussion in literature (Bakir et al., 2017; Ziccardi et al., 2016; Koelmans et al., 2016; Lohmann, 2017; Hartmann et al., 2017), but this was not considered in the present study. Our predictions about future ingestion assumed an ever-increasing plastic emission into the marine environment, but product innovation and policy actions in future decades could change those emission rates.

3.2. Effects of microplastics

The available effect data from literature, i.e. NOECs and LOECs for marine species exposed to microplastics, the size of the particles, the type of plastics used, and the endpoints used to assess the potential effects are summarised in Table 1 and further described in supportive information. A species sensitivity distribution (SSD) was constructed from the data (Fig. 2). The normal distribution was fitted to the log-transformed effect data. Using the method of Aldenberg and Jaworska (2000) (Aldenberg and Jaworska, 2000), an HC_5 of 33.3 particles L^{-1} (95% confidence interval: 0.36 particles L^{-1}–13,943 particles L^{-1}) was calculated. As the quality and quantity of the effect data incorporated in the SSD should be considered low (EU, 2006), and given the limited number of taxonomic groups representing only a few feeding strategies and trophic levels and the current lack of standard test methods, the highest AF (i.e. 5) was used on the HC_5, resulting in a predicted no effect concentration ($PNEC_{pelagic}$) of 6650 particles m^{-3} (95% confidence interval: 70–2.8×10^6 particles m^{-3}). Sediment effect data were highly underrepresented in literature and no SSD was constructed. Based on long-term effect data of Van Cauwenberghe et al. (2015) (Van Cauwenberghe et al., 2015) and by applying an AF of 1000 (in line with the European legislation) to the NOEC of the most sensitive endpoint (EU, 2006) (*Arenicola marina*: metabolic rate, 5.4×10^5 particles kg^{-1}) a PNEC for marine sediments ($PNEC_{benthic}$) of 540 particles kg^{-1} sediment was derived. As the present study is a pioneer work, the safe concentrations calculated in the present research come with some uncertainty and we acknowledge that additional adequate scientific research is needed to verify our findings. Of primary importance is the need for reliable long-term concentration – response effect data (at environmentally relevant

Table 1
Marine species (chronic) effect to microplastics data selected for the PNEC calculation. The most sensitive endpoint is provided for each species. Type of plastics used: polystyrene (PS), polyethylene terephthalate (PET), high density polyethylene (HDPE), polyvinyl chloride (PVC), polyethylene (PE). Numbers between brackets refer to the reference list.

Phylum	Species	Most sensitive endpoint	Type of plastic and size	NOEC *(particles mL^{-1})*	Reference
Ochrophyta	*Skeletonema costatum*	Growth	PVC 1 μm	3.4×10^6	Zhang et al. (2017)
Mollusca	Mytilus edulis	Metabolic rate	PS 10–90 μm	110	Van Cauwenberghe et al. (2015)
	Crassostrea gigas	Reproduction	PS 2–6 μm	96.8	Sussarellu et al. (2016)
	Ostrea edulis	Abundance and biomass	HDPE 0.48–316 μm	0.075	Green (2016)
	Perna viridis	Filtration and respiration rates	PVC 1–50 μm	6.0×10^5	Rist et al. (2016)
	Brachionus koreanus	Reproduction and life span	PS 0.05–6 μm	7.3×10^8	Jeong et al. (2016)
	Scrobicularia plana	Antooxidant capacity and DNA damage	PS 20 μm	2	Ribeiro et al. (2017)
	Pinctada margaritifera	Energy balance and gametogenesis	PS 6–10 μm	0.16	Gardon et al. (2018)
Arthropoda	*Tigriopus japonicus*	Mortality	PS 0.05–6 μm	2.1×10^5	Lee et al. (2013)
	Centropages typicus	Ingestion rate	PS 0.4–30.6 μ m	2000	Cole et al. (2013)
	Calanus helgolandicus	Feeding	PS 20 μm	37.5	Cole et al. (2015)
	Parvocalanus crassirostris	Reproduction	PET 5–10 μm	5000	Heinder et al. (2017)
Echinodermata	*Tripneustes gratilla*	Growth	PE 10–45 μm	100	Kaposi et al. (2014)
	Paracentrotus lividus	Fertility	PS	500	Martínez-Gómez et al. (2017)

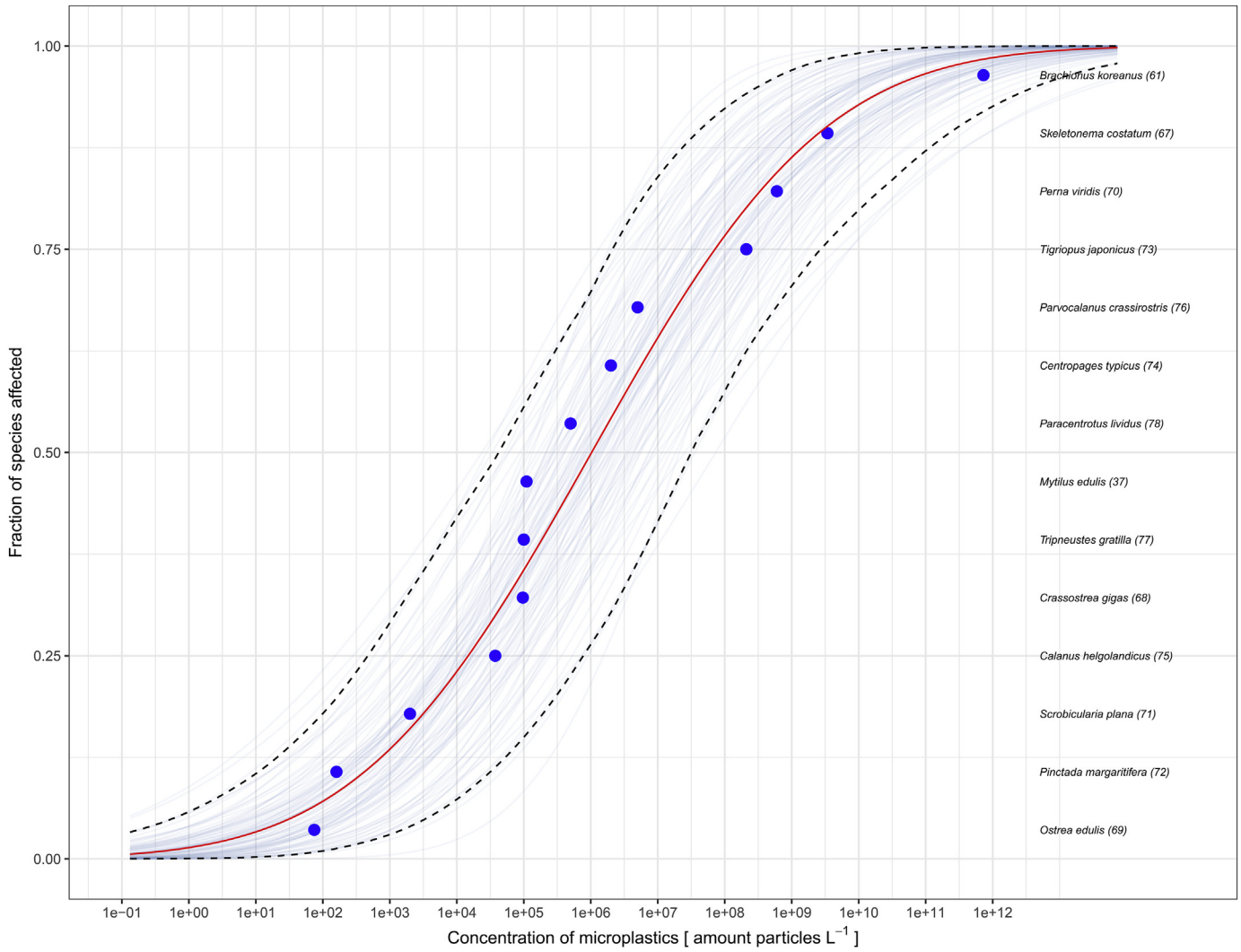

Fig. 2. Species sensitivity distribution (SSD) for buoyant microplastics (in particles L^{-1}). Chronic no observed effect concentrations (NOEC) and chronic lowest observed effect concentrations (LOEC) were derived using standard procedures (Table 1). If several chronic NOEC or LOEC values for different toxicological endpoints were available for a single species, the lowest value was used. Note that a log-normal distribution is fitted and that the labels indicate the species that was exposed to microplastics. Detailed information on the effect data from literature, i.e. marine species exposed to microplastics, 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 supportive information. Blue dots are the 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. Numbers between brackets refer to the reference list. (For interpretation of the references to colour in this figure legend, the reader is referred to the Web version of this article.)

microplastics concentrations) for a wider range of organisms representing different feeding strategies and more (and higher) trophic levels than presently available. The endpoints assessed should be ecologically relevant. So far, too much focus has been put on laboratory experiments in which a limited number of model species have been exposed to unrealistic high microplastic concentrations in poor test designs. In future years there is a clear need for adequate scientific data on the effect of microplastics for benthic organisms, a group that is currently underrepresented. A second aspect that needs further research is the size-dependent impact of microplastics. Recently, Jeong et al. (2016) found that smaller microplastics cause more effect. Indeed, in vitro tests revealed that anti-oxidant-related enzymes and mitogen-activated protein kinases signalling pathways were significantly activated in response to microplastic exposure in a size-dependent manner. However, due to the lack of sufficient effect data generated with a certain particles size (Table 1), no size dependent SSD could be made in the present work. Also, microplastics used in ecotoxicological research are often monodisperse or cover a narrow size range, and are generally much smaller than microplastics from exposure data. This discrepancy or miss match of size in 'exposure prediction (wide range from 0.001-5 mm)' and 'effect data (narrow size range)' adds some additional noise to the outcome of the SSD (Fig. 2). Regarding exposure assessment, complex transport mechanisms (both horizontal and vertical) of microplastics (Van Sebille et al., 2015; Eriksen et al., 2014; Kukulka et al., 2012; Fossi et al., 2017; Kooi et al., 2017), have not been integrated in our modelling approach. As such, we have considered that once a microplastic particle enters a particular environmental compartment (being pelagic, beach, or seabed) it remains in that compartment until it is completely fragmented. The assumption of non-existing transfer between environmental compartments is clearly violated for small microplastics that are denser than seawater. These microplastics have a relatively long pelagic phase, but will eventually end on the ocean seafloor (Kooi et al., 2017). Likewise, we assumed that microplastics composed of PP and PE remain floating during their entire life time. However, it was shown that biofouling to floating particles can make them denser than seawater, and thus sink to the ocean floor

(Kooi et al., 2017). Regional gradients of microplastic pollution were not explicitly included in our nonspatial ecological risk assessment. For an in depth discussion on this topic we refer to Eriksen et al. (2014) and Kooi et al. (2017), but future research should study and quantify the potential sinks of microplastics such as sediments (Claessens et al., 2011), marine biota (Van Cauwenberghe et al., 2015), sea ice (Obbard et al., 2014), and the deep sea environment (Woodall et al., 2014; Van Cauwenberghe et al., 2013) as well as the fluxes between them. In our exposure model, we considered the weight loss to be independent from the particles size. This assumption is debatable, as smaller particles are expected to mechanistically degrade faster than larger ones (Gewert et al., 2017). However, by using a fixed weight loss over the years, we assumed worst-case conditions, which is a common approach in ecological risk assessment.

3.3. Risk of microplastics

PECs from the exposure assessment and the PNECs determined in the effect assessment are combined into a risk characterisation ratio (RCR; Eq. (7)) to assess overall risks to the environment. As our $PEC_{t,float}$ remains lower than the corresponding PNEC, we foresee no eminent threat of microplastic pollution up to 2100 (Fig. 1A). Indeed, on average the exposure concentration in the upper layer of the pelagic compartment remains below the PNEC of 6650 particles m^{-3} (Fig. 1A). However, in areas that are heavily polluted with floating microplastic particles, such as for example in coastal waters (>100,000 particles m^{-3}) (Cozar et al., 2017) or in narrow straits such as the Queen Charlotte Sound in the NE Pacific Ocean (9200 particles m^{-3}) (Desforges et al., 2014), adverse effects can potentially occur (Fig. 1A). Sedimented microplastics are expected to exert adverse effects in future years (Fig. 1B and C). About 5% of the marine microplastics wash ashore (Sherrington, 2016), which will eventually result in environmental concentrations that are close to and even exceed the safe concentration in the second half of the 21st century. Even to date, adverse ecological effects are to be expected on some highly polluted beaches (Fig. 1B and C), especially at the high strandline were the amount of microplastics peaks. Plastic remediation projects can help at a local scale, but mentality shifts that prevent plastics from being produced, used and emitted into the environment, sustainable alternatives for plastics, and science-based policy guidelines on safe concentrations are urgently needed.

The presence of microplastics in the marine environment has been an issue of concern for over a decade now. A great number of studies conclude that microplastics are a threat or risk to these systems. However, these conclusions are mostly based on conjecture and inadequate data sets (i.e. ingestion data sets are only part of the risk-related research question). The environmental risk assessment of microplastics in the marine environment presented here (based on an order of magnitude estimation and not focused on the potential role of microplastics as chemical vectors) suggests, for the first time, that direct effects of free-floating microplastics pollution in marine environments are expected to occur at concentrations exceeding 6650 particles m^{-3} (Figs. 1A and 2). On average, the exposure concentration in the upper pelagic compartment remains below this safe concentration to date and up to 2100. However, this safe concentration has already been exceeded in a few highly polluted coastal environments. Hence, our efforts in terms of product innovation and policy actions should be stimulated and there is a clear need for continuing research. As human populations continue to grow, and our dependence on plastic does not change under a business as usual approach, we may expect a steady and substantial increase in microplastic concentrations in both the pelagic and benthic marine environment (Fig. 1) and the probability of causing adverse effects.

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Appendix A. Supplementary data

Supplementary data related to this article can be found at https://doi.org/10.1016/j.envpol.2018.07.069.

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