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Including impacts of microplastics in marine water and sediments in life cycle assessment

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ABSTRACT

Microplastics (MPs) pose a threat to marine ecosystems. When released, MPs first reach the water column, where they can be ingested by pelagic species. MPs can then reach marine sediments, a potential sink, where they may affect sediment-dwelling species. However, current life cycle impact assessment (LCIA) methods do not consider the impact of MPs in sediments, providing an incomplete picture when comparing environmental profiles of products and services.

This work builds on the MarILCA working group characterization factors (CFs) by computing updated physical effects on biota CFs that include both water and sediment compartments, as previous factors did not consider the latter. A simplified fate of MPs in the marine environment is modelled, combining fate in water and sediments and differentiating between MP polymers, sizes, and shapes. A combined exposure and effect factor for MPs in sediments (EEF_{sed}) is developed, calculated from a hazardous concentration for 20 % of species (HC20), derived from a species sensitivity distribution (SSD) of effect concentrations of 10 % (EC10) values. A methodology accounting for species feeding behaviour is proposed to derive ecosystem-level impacts via exposure through different compartments, expressed as the potentially affected fraction (PAF) of marine species.

Combining the fate, EEF_{sed}, and EEF_w (water) yielded updated marine CFs including impacts on both water and sediment-dwelling biota. CFs were tested in a textile LCA case study. Sediments were found to be a sink for high-density MPs, with EEF_{sed} (16 PAF m^3/kg) significantly lower than the previously reported EEF_w (1068 PAF m^3/kg). Developed marine CFs range from 34 to 5.4×10^8 PAF m^3 d/kg and are available for use in environmental decision-making.

1. Introduction

Microplastics (MPs), typically defined as plastic particles with a size comprised between 1 and 5000 μm, pose a threat to marine ecosystems (Woods et al., 2021). These small fragments originate from various sources, such as the breakdown of larger plastic items, microbeads from personal care products, tire road wear particles, or synthetic fibers shed from fiber-based textiles (Yang et al., 2021). When reaching the marine environment, MPs first go through the water column, where pelagic species can ingest them because of their small size. In most cases, although depending on density, shape and size, released MPs eventually reach the sediment layer, where they can be ingested by sediment-dwelling species (Van Cauwenberghe et al., 2015). Their ingestion by marine organisms can lead to detrimental health effects, with various effect mechanisms including physical damage or inhibited food assimilation (de Ruijter et al., 2020).

Life cycle assessment (LCA) is a tool to quantify the human interventions (extractions and emissions) associated with a product or service, over its life cycle, and their related potential impacts on ecosystem quality and human health (in addition to resource depletion/

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ecosystem services). However, life cycle impact assessment (LCIA) methods do not yet consider the impact of MPs in sediments (Corella-Puertas et al., 2023; Schwarz et al., 2024). This prevents decision-makers from obtaining an accurate portrait of the environmental footprint of products or services that emit MPs into the environment over their life cycle.

The MarILCA (MARine Impacts in LCA) working group was founded to develop methods for assessing marine litter impacts in LCA (Woods et al., 2021). Their efforts have led to characterization factors (CFs) which link the release of MPs to their impact on ecosystem quality (Corella-Puertas et al., 2023). An exposure and effect factor for the water compartment (EEF_w) was initially developed (Lavoie et al., 2022), later updated and combined with simplified environmental fate factors (FF) to obtain marine CFs (Corella-Puertas et al., 2022, 2023). However, these factors consider sediments as a sink without including the potential impacts which may occur in the sediments compartment once MPs reach them. This is due to the absence of EEF_{sed} in the sediments, which is an important gap as MPs often accumulate in sediments, where many benthic species live and feed (Van Cauwenberghe et al., 2015). More specifically looking at microfibers which are mostly high-density particles, the washing and handling of synthetic fibers are responsible for the release of 200 000 to 500 000 tons of MPs per year into marine ecosystems (European Environment Agency, 2022), making textiles one of the biggest sources of MPs in the marine environment (Acharya et al., 2021). To the best of our knowledge, only one study has attempted to conduct an LCA of a piece of clothing while considering the impacts of MPs. However, this study does not consider the effect of MPs in sediments (Salieri et al., 2021).

To address these gaps, this work builds on the MarILCA advances by computing updated characterization factors that consider the physical effects of MPs on water- and sediment-dwelling biota. To this end, this study aims to (i) introduce sediments into marine FF for MPs and (ii) develop a combined exposure and effect factor (EEF_{sed}) for MPs in the sediments compartment by reviewing effect concentration literature. Then, we (iii) combine FF, EEF_W and EEF_{sed} into updated marine midpoint and endpoint CFs for MPs. The last objective of this work is to (iv) illustrate the contribution of the impacts of MPs in marine water and sediments in LCA, by testing the developed CFs in an LCA case study on textiles.

2. Methods

2.1. Fate factors

Fate factors (FFs) quantify the distribution and time spent by MPs in each marine compartment. FFs and CFs developed in this work only consider the continental scale impacts (coastal marine water) as more than 90 % of marine species are found at this scale (Tittensor et al., 2010). Additionally, the volume of global marine water is multiple orders of magnitude higher than that of continental marine water, making the dilution volume for MPs that reach the global scale much greater, overall affecting fewer species. Ongoing work by the MariLCA group aims to characterize the transfer rates of MPs between the continental and global scales (Hajjar et al., 2023), however, impacts at the global scale are assumed negligible until better data becomes available. Corella-Puertas et al. (2023) consider the fate and effect in one marine compartment (*i.e.* the water column). The following loss and transfer processes rate constants, in each compartment, are required to add the sediment compartment to the fate:

- the **degradation rate** (k_{deg}). The degradation rate of each polymer was calculated based on the specific surface degradation rates (SSDR) (Corella-Puertas et al., 2023; Maga et al., 2022). It was assumed that the SSDR in the sediments was equal to the rate in the water column due to the lack of sediment-specific SSDR data.
- the **deep burial rate** ($k_{deep\ burial}$), which represents the rate at which MPs reach deeper levels in the sediments where they are assumed to exit the marine environment and assumed to become inaccessible to sediment-dwelling species (Hajjar et al., 2023; Quik et al., 2023)
- the **resuspension rate** ($k_{resuspension}$) characterizes the transfer of MPs from the sediments back into the water column (Hajjar et al., 2023; Quik et al., 2023)
- and the **sedimentation rate** ($k_{sedimentation}$) characterizes the rate of transfer of MPs from the water column to the sediments which depends on the density of MPs (Corella-Puertas et al., 2023).

All rate constants are expressed in [yr^{-1}]. Considering the two compartments, a rate constant matrix **K** and a fate factor matrix **FF** were constructed following the USEtox approach (Fantke et al., 2017; Rosenbaum et al., 2007; Appendix A). Rate data and calculations for each polymer can be found in Appendix C.

2.2. Sediments exposure and effect factor

Exposure and Effect Factors (EEFs) link the fate of pollutants to their potential impacts on ecosystem quality. In this section, an EEF_{sed} is developed for MPs, in the sediments compartment.

2.2.1. Data collection

A literature review was done to collect effect data of MPs on benthic organisms, with exposure via sediments to later compute the EEF_{sed}. Keyword search details can be found in Appendix A. Data for both freshwater and marine sediment-dwelling species were collected, as insufficient data points specific to the marine environment could be gathered. Moreover, existing methods for evaluating ecotoxicity in LCIA do not differentiate between freshwater and marine species in effect factor developments (Fantke et al., 2018; Owsianiak et al., 2023). This is mainly due to the lack of marine species-specific effect data for the wider range of pollutants; where marine ecotoxicity is considered as a separate indicator in available LCIA methods, it is usually based on applying 1:1 effect test data from freshwater species (Verones et al., 2020). Studies with an exposure through water were excluded to keep only tests done with organisms exposed to MPs through sediments. Additionally, only experiments done with virgin polymers were kept in order to focus on the physical effects on biota of MPs and separate them as much as possible from the toxicological effects caused by polymer additives. A total of 29 articles matching these criteria were gathered, and all experimental data resulting in effect data (i.e. Lowest Observed Effect Concentration (LOEC), No Observed Effect Concentration (NOEC), and Effective Concentration (ECx)), reported as weight or count of MPs per weight or volume of sediments were kept. Each data point, the species tested, the polymer type, size and shape, as well as the concentration and biological endpoints studied were collected in an Excel sheet (Appendix B). The biological endpoints considered were reproduction, survival, growth (as recommended in USEtox) as well as feeding. Further, when multiple biological endpoints were tested in an experiment, each value obtained was kept as a separate data point.

2.2.2. Exposure and effect factor calculation

Data gathered was used to compute the EEF_{sed}. To comply with global recommendations for ecotoxicity impacts in LCIA, the EEF_{sed} is calculated from a hazardous concentration of 20 % (HC20), derived from a species sensitivity distribution (SSD) of effect concentrations of 10 % (EC10) (Owsianiak et al., 2023). This method requires data points for at least 5 different species, spanning 3 different taxonomic groups. When EC10 values were not reported in a study, extrapolation factors obtained from Aurisano et al. (2019) were used to convert NOECs and LOECs gathered in the literature (Section 2.2.1) to EC10eq. The extrapolation factors used are given in Table S2 of Appendix A. Further, when EC10 data was available for a particular species, LOECs and NOECs data for that species were discarded to minimize additional uncertainty associated with the use of extrapolations to EC10eq. Therefore, EC10eq derived from LOECs and NOECs were used in the SSD only when EC10 data was unavailable for a given species.

A log-normal fitted SSD curve was generated using the ssdtools package in R, allowing for a visual representation of the EC10eq data obtained for each species and taxonomic group. The $HC20_{EC10}$ was obtained from the curve and converted from g_{MPs}/kg_{DW} (DW: dry weight of sediments) to kg_{MPs}/m^3_{WS} (WS: wet sediments) before computing the EEF_{sed}. The conversion was done using the bulk density of wet sediments (Fantke et al., 2017) and the average sediments wet/dry ratio (Van Cauwenberghe et al., 2015; Appendix A). The units of EEF_{sed} and EEF_w are $\left[PAF_{sed}\ m^3_{sed}/kg_{in\ compartment}\right]$ and $\left[PAF_w\ m^3_w/kg_{in\ compartment}\right]$. The methodology described by Lavoie et al. (2022) was used to compute the 95 % confidence interval (CI) around EEF_{sed}. The calculations of the EEF and CI can be found in Appendix B.

2.3. Scaling of impacts via exposure compartment

To develop a characterization factor (CF) representing impact potentials on the marine ecosystem as a whole, where equal relevance is given to water and sediments-dwelling species, the compartmental impacts in the water column and sediments were each scaled from a potentially affected fraction of water- or sediment-dwelling species (PAF_w or PAF_{sed}, respectively) to a potentially affected fraction of overall marine species (PAF_{mar}). In other words, a scaling approach is proposed, allowing to derive overall marine ecosystem-level impact characterization factors via exposure in different marine compartments.

The scaling is done by multiplying each compartmental impact (water and sediments) by the percentage of marine species that are exposed via the corresponding marine compartment (water- or sediment-dwelling species). Two scenarios of exposure were considered: first, a division based on the **fraction of species living** in each corresponding marine compartment, characterized using the marine zones (i.e. pelagos or benthos) species richness. Second, a division based on the **fraction of species feeding** in the corresponding compartment, characterized using the feeding behaviours of marine species. These two approaches cannot be combined or considered equivalent as, for some species, the compartment in which they live could be different than the compartment in which they feed. Moreover, double counting of species is avoided in both approaches as the databases from which data was gathered only contain one entry per species, regardless of their life stage. Both approaches are explained and tested, before selecting one scaling method in section 3.3.1 of the results.

2.3.1. Marine zones species richness

This first scaling approach is based on the compartment in which marine species spend the majority of their time. This approach assumes that a species is exposed in the compartment in which it mostly lives, for example, exposure in the sediment zone via skin contact or gills but does not consider where that species feeds.

Data gathered in the literature on marine zones species richness yielded the following distribution of marine species: 8 % of pelagic species, 63 % of epibenthic species and 29 % of endobenthic species in continental seawater (Table S3 and Figure S1 of Appendix A) (Ahyong et al., 2023; Archambault et al., 2010; May and Godfrey, 1994; Widdicombe and Spicer, 2008). This distribution allows for a differentiation of the medium through which organisms are exposed to MPs, hypothesizing that organisms are affected by MPs by spending part of their lifetime in a specific compartment. We assumed that pelagic species are only exposed through water, whereas endobenthic (or infaunal) species are only exposed through sediments as they live within the sediments. On the other hand, epibenthic species can be exposed through either the water column or sediments, as they are mobile organisms that can move between the sediments and water column compartments (Sivapriya et al., 2022). This approach thus considers that epi-benthic species can be exposed through both water and sediments. Note that it was not necessary to divide the pelagic region into smaller ecological zones (e.g. epipelagic, mesopelagic, etc), as all species in the pelagic region are assumed to be only affected by MPs by exposure via water in this approach.

2.3.2. Feeding behaviour

This second scaling of impacts is based on the compartment in which marine species feed. This approach assumes that MPs adversely affect marine species with an effect mechanism linked to their ingestion, disregarding other exposure types (e.g. through gills or skin contact). This assumption is discussed and rationalized in section 3.3.1 of the results.

The WORMS database contains data on the number of *known* species belonging to each feeding behaviour (Ahyong et al., 2023). Each feeding behaviour is associated with an exposure pathway: water or sediments. When no clear trend could be identified for a specific feeding behaviour, it was assumed that half the species belonging to this feeding type feed in water and the other half in sediments. In fact, the species belonging to these feeding behaviours were not found to predominantly feed in one compartment or the other, an equal split was, therefore, the most reasonable assumption. Table 1 shows each feeding behavior and the associated exposure compartment. Sediments were determined to be the main exposure pathway for deposit and detritus feeders, as species belonging to these feeding behaviour types feed on sediments, thus frequently ingesting sediment particles (Snelgrove, 2013). Additionally, most species included in the EEF_{sed} belong to the deposit and detritus feeding behaviours, which supports this decision.

Further, it was hypothesized that filter and suspension feeders were mainly exposed through water, as these organisms feed by filtering large amounts of water. Overall, this results in 44.6 % of marine species feeding through water (74438 known species) and 55.4 % through sediments (92549 known species). It was assumed that this ratio was representative of the relative number of *existing* species belonging to the

Table 1
Feeding behaviour data and exposure compartment.

Feeding behaviour (Ahyong et al., 2023)	Number of known species (Ahyong et al., 2023)	Exposure compartment (hypothesis)	Species that belong to this feeding type, in EEF_{sed}
Carnivore	1065	50 % sediments/ 50 % water	
Deposit feeder	18 335	sediments	*Arenicola marina, Oncholaimus campylocercoides, Lumbriculus variegatus, Tubifex* spp., *Potamopyrgus antipodarum, Ennucula tenuis, Abra nitida, Hediste diversicolor*
Detritus feeder	29 632	sediments	*Asellus aquaticus, Chironomus tepperi (larvae), Chironomus sancticaroli, Chironomus riparius, Hyalella azteca, Gammarus pulex*
Filter feeder	18 921	water	*Sphaerium corneum, Caenorhabditis elegans*
Grazer	48 384	50 % sediments/ 50 % water	*Hyalella azteca*
Omnivore	1870	50 % sediments/ 50 % water	–
Parasitic	47 626	N/A	–
Predator	36 249	50 % sediments/ 50 % water	–
Scavenger	2662	50 % sediments/ 50 % water	–
Suspension feeder	10 935	water	–

ten feeding behaviour types reported for the marine environment, as the ratio of *known* to *existing* species is likely similar for every feeding behaviour.

2.3.3. Species distribution factor

The compartmental CFs are expressed as a potentially affected fraction of water- and sediment-dwelling species, respectively, integrated over the respective exposure volume of water ($PAF_w\ m_w^3$) and sediments ($PAF_{sed}\ m_{sed}^3$), and are integrated into PAF of overall marine species over a total marine environment volume ($PAF_{mar}\ m_{mar}^3$). This was done using the percentage of species exposed through each compartment, as well as the respective volume of each compartment. The fraction of species affected via each compartment, for each approach, is described in sections 2.3.1 and 2.3.2. Both approaches were tested in order to compare the results. To represent the scaling of compartmental impacts in a matrix form, we introduce the species distribution factor matrix (**SDF**) which can be used to scale compartmental CFs to obtain CFs at the overall marine ecosystem level. The $\mathbf{SDF}_k$ is defined with the following structure, for ecosystem *k*, which comprises *n* compartments:

$$\mathbf{SDF}_k = \begin{bmatrix} SDF_1 & SDF_2 & \ldots & SDF_n \end{bmatrix} \tag{1}$$

Where SDF_n is the percentage of species of ecosystem k affected via compartment n. In the marine ecosystem with water and sediments compartments, the following SDF matrix is obtained:

$$\mathbf{SDF}_{\mathbf{mar}} = \begin{bmatrix} SDF_w & SDF_{sed} \end{bmatrix} \tag{2}$$

In future studies, the marine ecosystem could be further disaggregated if additional CFs for MPs are determined for specific marine zones. For example, the pelagic zone could be split into water surface and water column, as identified by Hajjar et al. (2023).

2.4. Characterization factors

Characterization factors (CFs) link the emission of MPs to their potential impacts on ecosystem quality as an area of protection. A combined exposure and effect factors diagonal matrix, **EEF**, was set up with the EEF_{sed} and EEF_w on the diagonal. These EEFs are expressed as a potentially affected fraction of water- or sediment-dwelling species over a volume of water or sediments, respectively ($PAF_w\ m_w^3$ or $PAF_{sed}\ m_{sed}^3$). The compartmental CFs matrix, at midpoint level, can be obtained with the following equation:

$$\mathbf{CF}_{comp,mar} = \mathbf{EEF}\ \mathbf{FF} = \begin{bmatrix} CF_{w,w}\ [PAF_w\ m_w^3\ d/kg] & CF_{w,sed}\ [PAF_w\ m_w^3\ d/kg] \\ CF_{sed,w}\ [PAF_{sed}\ m_{sed}^3\ d/kg] & CF_{sed,sed}\ [PAF_{sed}\ m_{sed}^3\ d/kg] \end{bmatrix} \tag{3}$$

Where **FF** is the fate factor matrix. The units of $\mathbf{CF}_{comp,mar}$ are shown in equation (3) (Bulle et al., 2019; Owsianiak et al., 2023) . This results in a two-by-two matrix, $\mathbf{CF}_{comp,mar}$, where the left-hand side corresponds to compartmental CFs for an MP emission in the water column and the right-hand side to an emission in sediments (in the case of fragmentation of MPs from macro-plastics within the sediments). The rows represent the receiving compartments in which impacts occur.

Each row is then divided by the volume of each marine compartment, to remove the volume over which impacts occur and retrieve the PAF in each compartment. Indeed, impacts ($PAF_i\ m_i^3\ d/kg$) are integrated over a volume of compartment using the EEF_i and cannot be summed with impacts in another compartment (obtained with another EEF) without considering the different respective compartment dilution volumes. Further, the **SDF** developed in this work is used to go from a potentially affected fraction of water or sediments-dwelling species (PAF_w or PAF_{sed}) to a PAF of overall marine species (PAF_{mar}) while aggregating compartmental impacts for the same compartment of emission. Finally, the CFs are multiplied by the marine volume to obtain a PAF of marine species over the volume of the marine ecosystem.

$$\begin{aligned} \mathbf{CF}_{\mathbf{midpoint,mar}} &= V_{mar}\ \mathbf{CF}_{\mathbf{eco,mar}} \\ &= \begin{bmatrix} CF_w\ [PAF_{mar}\ m_{mar}^3\ d/kg] & CF_{sed}\ [PAF_{mar}\ m_{mar}^3\ d/kg] \end{bmatrix} \end{aligned}$$

Where $V_{mar} = V_w + V_{sed}$ and where the left-hand side corresponds to ecosystem-level CFs for an MP emission in the water column and the right-hand side to an emission in sediments, which are both expressed as $[PAF_{mar}.m_{mar}^3.d/kg]$. Detailed matrix manipulations and the Python code used to compute CFs and their 95 % confidence interval can be found in Section 1.3 (matrix) and Section 3 (code) of Appendix A.

2.4.1. Midpoint to endpoint conversion

LCIA method Impact World + uses $[PDF\ m^2\ yr/kg_{emitted}]$ units at damage level (Bulle et al., 2019). The conversion from mid-to endpoint was done using the following:

$$\mathbf{CF}_{endpoint,mar} = \frac{SF}{365\ d \cdot \text{compartment depth}} \mathbf{CF}_{midpoint,mar} \tag{4}$$

Where SF is the severity factors, in units of [PDF /PAF], which allows for the conversion from PAF to PDF (Bulle et al., 2019). The value used is 1 PDF/PAF, following the GLAM recommendation for EFs based on an $HC20_{EC10}$ (Oginah, 2023). The compartment depth, expressed in [m] allows for the generic conversion from m_{mar}^3 to m_{mar}^2. The value used was 100.1 m, as the continental seawater depth, where MPs are potentially affecting species, is 100 m (Corella-Puertas et al., 2023; Fantke et al., 2017) and the sediments compartment has a depth of 10 cm (Hajjar et al., 2023). In this case, the matrix operations and conversion to endpoint is equivalent to multiplying $\mathbf{CF}_{\mathbf{eco,mar}}$, expressed as $[PAF_{mar}\ d/kg]$, by the global 'continental seawater' area (and converting PAF.d to PDF.yr), which equals the sum of each continent's seawater area (Fantke et al., 2017).

The Global Guidance for Life Cycle Impact Assessment Indicators and Methods (GLAM) method uses $[PDF\ yr/kg_{emitted}]$ units at damage level (UNEP, 2019). Similarly, a conversion was done using the global volume of marine coastal zones from USEtox, which corresponds to the sum of coastal zones volumes for all continents and equals 2.20E+15 m^3 (Fantke et al., 2017).

$$\mathbf{CF}_{endpoint,mar-GLAM} = \frac{SF}{\text{global volume of marine coastal zones}}\ \mathbf{CF}_{midpoint,mar} \tag{5}$$

2.4.2. Materials included

Corella-Puertas et al. (2023) calculated CFs for 11 polymers. In this study, two (2) additional polymers, namely polyurethane (PU, spandex) and polyacrylonitrile (PAN, acrylic), are included in the dataset, in addition to the polymers studied by Corella-Puertas et al. (2023). Further, four (4) natural and cellulose-based materials were added: cotton, viscose, linen and wool. This brings the total of materials for which CFs are computed to 17.

We also compute characterization factors for cellulosic based natural and semi-synthetic fibers (e.g. cotton, linen, viscose), as they can also induce adverse physical effects on biota (Walkinshaw et al., 2023). The values of the EEFs in water and sediments were taken as 50 % of the polymer EEFs for natural and semi-synthetic materials, as insufficient data was found to develop EEFs specific to these cellulosic fibers so far. FFs were developed following the same methodology as for polymers, hence including specific sedimentation rates (based on density) and degradation rates (based on the surface-specific degradation rate). Several studies have indeed shown an adverse effect on living organisms from cotton and regenerated cellulose-based fibers exposure, such as viscose, with an effect smaller than that of MPs (Kim et al., 2021; Siddiqui et al., 2023; Walkinshaw et al., 2023). Walkinshaw et al. (2023) have shown that cotton microfibers induce a decrease in growth of mussels of about 50 % of the one observed with the same concentration of polyester MPs.

2.5. Case studies

The CFs developed in this study were tested in a sportswear textile case study. Primary data was obtained from an industrial partner based in Québec. The goal of the study was to demonstrate the relevance of including the impacts of MPs in marine sediments in LCA of microplastic-emitting products and services, as well as to put the physical effects on biota of MPs in perspective with the impacts of other chemicals as well as other impact categories commonly used in LCIA(e. g. climate change).

The developed methodology was also tested on two textile case studies of the UNEP report entitled "Sustainable and circular textile value chains: Insights from life cycle assessments". In both case studies, the original LCA results did not include the physical effects on biota of emitted MPs. Hence, the goal of the work done for the UNEP report was to add the impacts of MPs. The inventory and calculations can be found in Appendices G and H.

2.5.1. Description and product boundary system

Three different pairs of leggings are compared in this study: the first one is 80 % polyamide (PA or Nylon) and 20 % polyurethane (PU or Spandex), the second one 80 % polyester (PES) and 20 % PU, and the last one is 80 % recycled PES (REC-PES) and 20 % PU. The functional unit chosen for the case study is "To wear a pair of leggings 70 times, in Québec". A number of use of 70 was selected to represent the average number of times a pair of leggings is worn over its lifetime according to the PEFCR Apparel and footwear (PEFCR, 2022). The three pairs of leggings are assumed to have the same lifetime and to be worn and washed the same number of times. The system is cradle-to-grave and represents all the stages over the complete life cycle of the leggings, for all the times the garments will be worn. The boundary system of the pair of leggings includes the production stage of the raw materials (e.g. polyurethane, nylon and polyester), and their transformation into fibers in China. These steps are followed by the yarn spinning, the circular knitting of the yarn to make fabric, as well as the fabric dyeing, cutting, and sewing into a pair of leggings, in Québec. Further, the boundaries also include the use phase which includes washing and drying of the garment. Finally, the landfilling of the pair of leggings at its end-of-life is also part of the system. Transportation between various life cycle stages is also included. Graphical product system boundaries can be found in Figure S2 of Appendix A.

2.5.2. Life cycle inventory and microplastics pre-fate

An OpenLCA model was built for the product system described in section 2.5.1. Data was gathered from the literature and the industrial partner for each of the life cycle stages (Table S5 of Appendix A). A complete LCI can be found in Appendix D. Emissions of MPs over the life cycle of clothing originate from various life cycle stages, namely the fiber production, the yarn spinning, the circular knitting, the fabric dyeing, the manufacturing, the use phase (washing), and the end-of-life. The release rates of MPs for the production and use phases were estimated using data from the Plastic Footprint Network (PFN) (Earth Action, 2023). For the end-of-life stage, MPs fragmentation from mismanaged textiles was calculated using a fragmentation percentage reported by Pinlova and Nowack (2023) as well as mismanagement rates for macroplastics from the PFN (Earth Action, 2023). Detailed calculations on the MPs leakage can be found in Appendix D. A sensitivity analysis was done on the textile mismanagement rate at its end-of-life, as it varies greatly from one country to another. This was done to study its effect on the overall impact of the garment.

2.5.3. Software and LCIA method

OpenLCA v.2.0.2 was used to compute the LCI and LCIA results. The impact assessment method chosen is Impact World + (IW+) v2.1 with *Expert* and *Midpoint* versions (Bulle et al., 2019), which links the LCI results to damage at the midpoint and endpoint level. All impact categories of the methods with damage on ecosystem quality (EQ) were considered.

3. Results

3.1. Fate factors

Fate factors in the marine environment were computed for 17 different materials, 3 shapes and 5 sizes. The fate in the marine environment ranges from a few days (small, fast degrading particles) to thousands of years (large, slow degrading particles). Fig. 1 shows the fate of 10 μm microfibers.

For an emission in the water column, the fate of low and medium-density polymers is much longer in the water column than in sediments. This is due to their slow sedimentation compared to their degradation rate. On the other hand, high-density materials have a larger fate in sediments than in water, mainly caused by their faster sedimentation. However, this is not the case for polyhydroxyalkanoate (PHA), Viscose and Cotton, which are also high-density materials. This is because their degradation rate is higher, meaning that these fibers will spend less time in the sediments than the non-biodegradable high-density fibers, because they will (at least partly) degrade before reaching the sediments. Polyvinyl chloride (PVC) and Acrylic have significantly larger fates in the marine environment than other materials, as their degradation rates are the smallest of the dataset (Appendix C).

3.2. Sediments exposure and effect factor

3.2.1. Overview of collected data

For marine sediments, 18 data points were obtained, representing 7 species over 3 phyla and 2 taxonomic groups. For freshwater sediments species, 36 data points were identified, spanning 13 species over 5 phyla and 3 taxonomic groups. EC10s for three (3) species were obtained in the literature and extrapolations were required from NOECs and LOECs to EC10 for the other species for which EC10 data was not directly available. The number of taxonomic groups represented in the marine EEF_{sed} does not meet the USEtox requirements for an EEF based on a $HC20_{EC10}$, thus, a combined marine and freshwater EEF was computed to meet the recommendations. This combined marine and freshwater approach is also supported by the uncertainty range of the $EEFs_{sed}$ obtained for each aquatic environment (Appendix B). In fact, the combined marine and freshwater EEF_{sed} is within the uncertainty of both the freshwater EEF_{sed} and marine EEF_{sed}. Table S7 of Appendix A shows a summary of the effect data gathered in the literature.

3.2.2. Hazardous concentration and species sensitivity distribution

The $\mathbf{EEF_{sed}}$ obtained for the sediments is 16.2 [1.7–152.9] PAF m^3/kg, significantly smaller than the value for the water column of 1067.5 [358.1–3182.1] PAF m^3/kg (Corella-Puertas et al., 2023). This EEF_{sed} was obtained from the $HC20_{EC10}$ concentration retrieved from the SSD curve ($HC20_{EC10}$ = 0.0125 g/kg, Fig. 2). These EEFs are

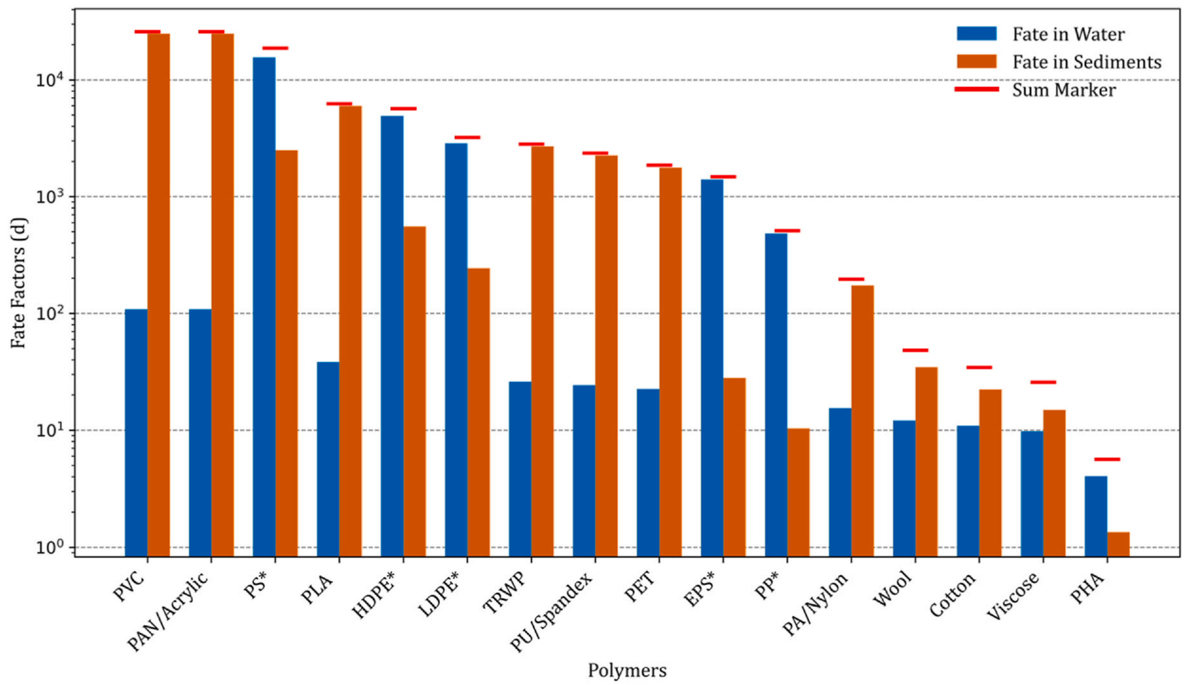

Fig. 1. Fate factors (FF) in the marine environment for 10 μm microfibers, for an emission in marine water, shown on a log-scale. The * symbol designates low and medium-density polymers. The sum represents the total residence time of particles in the marine environment (water and sediments combined). PVC: Polyvinyl Chloride, PAN: Polyacrylonitrile, PS: Polystyrene, PLA: Polylactic Acid, HDPE: High-density polyethylene, LDPE: Low-density polyethylene, TRWP: Tire and road wear particles, PU: Polyurethane, PET: Polyethylene terephthalate, EPS: Expanded polystyrene, PP: Polypropylene, PA: Polyamide, PHA: Polyhydroxyalkanoate.

applicable to all types of micro- and nanoplastic as insufficient data is available to calculate EEFs specific to the size, polymer, or shape. Further, no statistically significant differences were found when enough data was available to compute size-, polymer- or shape-specific EEFs (Lavoie et al., 2022). However, further studies should investigate this once more experimental data is available.

Chironomus tepperi and *Chironomus sancticaroli* are the most sensitive species in the dataset. It is worth noting that all nematodes are in the lower half of EC10eq values, indicating higher sensitivity to MPs, while both angiosperms (*Elodia* sp and *Myriophyllum spicatum*) are in the top half (Fig. 2). No clear trend is seen for other phyla (Annelids, Arthropods and Molluscs). The SSD integrates EC10 data from species at multiple life stages (larvae, juvenile and adult), therefore considering their life stage-dependent sensitivity to MPs.

3.3. Marine characterization factors

3.3.1. Scaling approach

The marine physical effects on biota CFs obtained using both approaches (Living compartment, section 2.3.1, and feeding compartment, section 2.3.2) are in the same order of magnitude (Figure S3, Appendix A). However, CFs computed with the second scaling method (i.e. Feeding behaviour) are smaller than with the first one (i.e. Marine zones) as this first scaling approach considers some species to be exposed through both water and sediments compartments, which increases the overall CFs (Section 2.3.1).

While data gathered in the literature (Section 2.3.1) revealed that most marine species are benthic (both epi- and endobenthic) and thus mostly live within or on top of sediments (Ahyong et al., 2023; Archambault et al., 2010; May and Godfrey, 1994; Widdicombe and Spicer, 2008), this does not necessarily reflect the main exposure pathway of organisms to MPs. A species could for example reside within sediments, while feeding in water, thus being mainly exposed through water. Further, various studies have shown the importance of feeding ecology, particularly feeding behaviour, on exposure to MPs (D'Avignon et al., 2023; Mizraji et al., 2017; Scherer et al., 2017). Additionally, de Ruijter et al. (2020) have reviewed 105 MPs effect studies on aquatic species and investigated the suggested and/or demonstrated effect mechanisms of MPs whether exposed via water or sediments. Their findings show that the two most suggested and demonstrated effect mechanisms are "inhibited food assimilation and/or decreased nutritional value" as well as "internal physical damage", which are both due to ingestion of MPs that occur through feeding. External damage, linked to the compartment in which species feed, has also been suggested/demonstrated as an effect mechanism in some effect studies but to a lesser extent. This highlights feeding as the main exposure pathway for living organisms and supports a scaling of impacts from the compartmental level to the ecosystem level based on where species feed, rather than where species live. The feeding behaviour scaling approach was therefore chosen.

3.3.2. Characterization factors results

EEF was combined with **FFs** (equation (8)) to obtain midpoint marine physical effects on biota CFs ($PAF_{mar}\ m^3_{mar}\ d/kg$) for various materials, shapes and sizes and are available in Appendix E (Fig. 3).

Midpoint CFs, expressed as $kg_{PP,1\mu m,sphere\ eq}$ were also calculated and proposed as an alternative midpoint units using 1 μm microspheres polypropylene (PP) as a reference. This midpoint was computed by dividing each endpoint CF by the CF of 1 μm microspheres PP particles, for an emission in marine water. This reference was chosen as it is a common small-sized particle emission. Endpoint CFs (and their associated 95 % confidence interval) are also available in Appendix E for units compatible with the IMPACT World + method [PDF m^2 yr /$kg_{emitted}$] and ReCiPe [species yr/$kg_{emitted}$], as well as in Appendix F for GLAM compatible units [PDF yr/$kg_{emitted}$].

Higher CFs for low and medium-density materials were obtained for larger particles, which is due to their higher fate (i.e. high persistence) in the water compartment (Fig. 4) and high EEF_w. On the other hand, the large fate in sediments (for high-density, large particles) is also a driver of high CFs (Fig. 4). In fact, while exposure and effect (EEF_{sed}) are smaller in this compartment, its small dilution volume makes potential

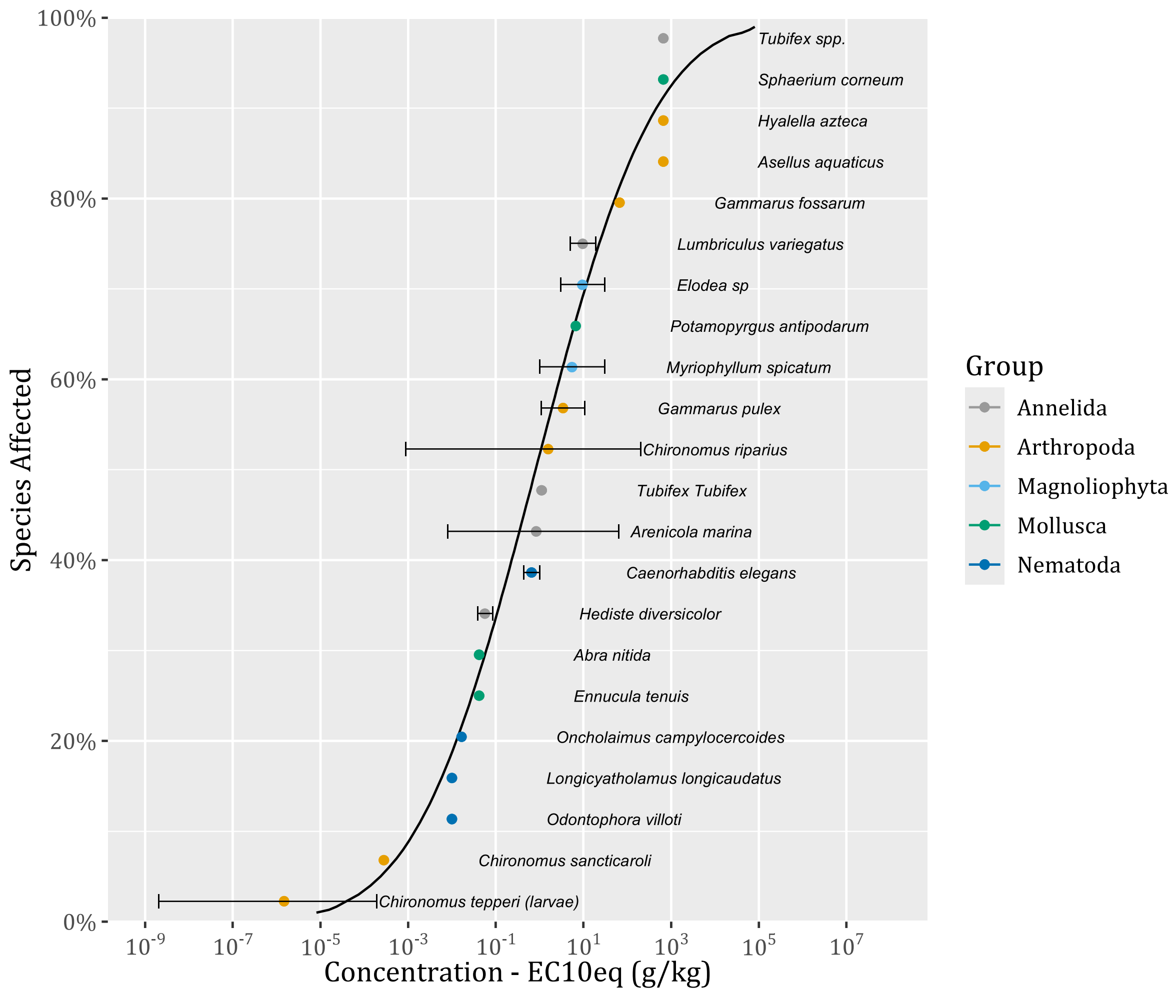

Fig. 2. Species Sensitivity Distribution (SSD) for freshwater and marine species exposed to microplastics through sediment. Each dot represents the average EC10eq for a specific species. Error bars show the minimum and maximum value of the EC10eq, when multiple values were obtained in the literature for a particular species.

impacts on sediment-dwelling biota significant for high-density polymers. Therefore, as the EEFs (water and sediments) are applicable to any polymer, the potential impacts (CFs) are only a function of the fate in water and sediments of the different materials, shapes and sizes.

3.4. Case studies

3.4.1. UNEP report on textile

The CFs developed in this study were tested in two textile case studies of the UNEP report entitled “Sustainable and circular textile value chains: Insights from life cycle assessments” (Section 2.5). The results, which can be found in Appendices F and G, are discussed in the report.

3.4.2. Ecosystem quality damage

Physical impacts on biota represent between 0.42 % (Polyamide/Polyurethane textile) and 1.48 % (Recycled Polyester/Polyurethane textile) of the total damage on ecosystem quality (EQ), considering all impact categories of Impact World+, for a use phase in Québec, Canada (Fig. 5). This is partly explained by the fact that MPs may not reach marine ecosystems in high-income countries because of the high-efficiency filtration in wastewater treatment plants (WWTP). Indeed, MPs rather end up in agricultural soils when WWTP sludges are land-applied on fields (Rolsky et al., 2020). The PFN estimates that 38 % of MPs emitted into wastewater, in Canada, are released into terrestrial ecosystems while only 13 % end up in marine environments (Earth Action, 2023). Calculations and release rates are shown in Appendix A. In the Québec context, the recycled polyester and polyurethane (RES-PES/PU) pair of leggings has the lowest damage on EQ.

A use phase and end-of-life in a low-income country (LIC) was chosen as a sensitivity analysis by changing the release rates of MPs during the use phase, as well as the mismanaged rate of textiles (Section 2.5.2). The results of the sensitivity analysis show that in an LIC country, the impacts of MPs would represent 7.6 %–22.2 % of the total damage on ecosystem quality, due to the higher leakage rate and mismanagement rate of textiles (Fig. 5).

3.5. Recommendations for LCA practitioners

Based on the default MPs sizes identified by Corella-Puertas et al. (2023) for microspheres/fragments, microfibers and microfilms, default CFs are proposed in Table 2 for LCA practitioners that don’t have access to full information on the MPs leaked over the life cycle of the studied product or service. It is however still recommended to use specific CFs (Appendix E) when information on the material, size and shape is available to the practitioners. To use these CFs, a detailed inventory should be developed using existing methodologies that quantify MPs leakage over the life cycle of products and services, such as the PFN (Earth Action, 2023), or using primary data from suppliers and laboratory tests.

Further, the CFs on 1 μm MP (Appendix E) are proposed as a proxy for nanoplastics, as done and rationalized by Corella-Puertas et al.

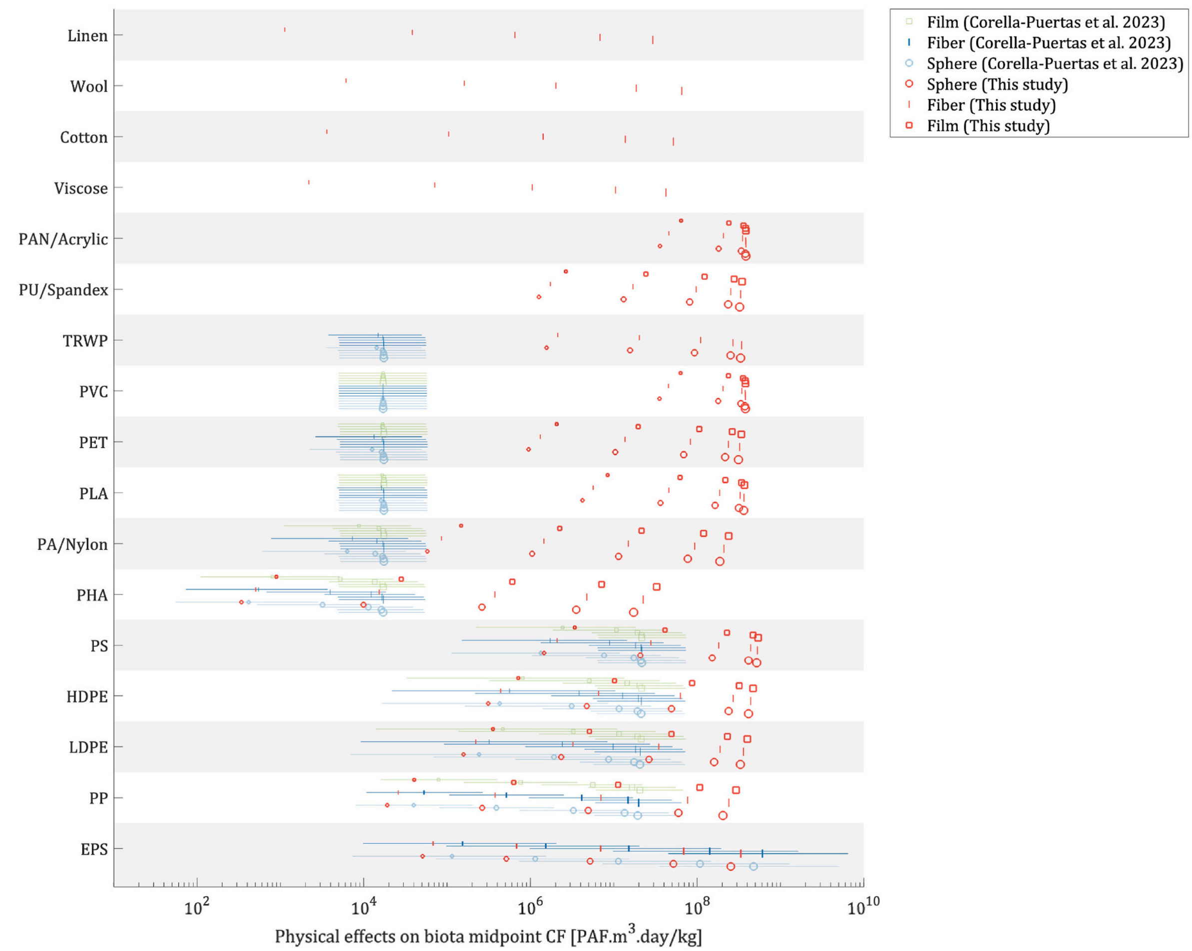

Fig. 3. Marine physical effects on biota midpoint CFs of films (squares), fibers (lines) and spheres (dots) microparticles, for an emission in marine water. The microplastic size (1, 10, 100, 100, 5000 μm) is represented by the marker size (e.g. smallest marker = 1 μm micro plastic). Values in blue and green are the CFs computed by Corella-Puertas et al. (2023), and their corresponding 95 % confidence interval (horizontal bars). Values in red come from this study with changing marker sizes and shapes, underlying data can be found in Appendix E. PVC: Polyvinyl Chloride, PAN: Polyacrylonitrile, PS: Polystyrene, PLA: Polylactic Acid, HDPE: High-density polyethylene, LDPE: Low-density polyethylene, TRWP: Tire and road wear particles, PU: Polyurethane, PET: Polyethylene terephthalate, EPS: Expanded polystyrene, PP: Polypropylene, PA: Polyamide, PHA: Polyhydroxyalkanoate. (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

(2023). Caution should be exercised when using CFs of natural and semi-synthetic microfibers (cotton, viscose and wool) as an EEF specific to these particles has not yet been developed, and its value was so far assumed to be 50 % of that of MPs (Section 2.4.2).

4. Discussion

4.1. Fate factors

The overall residence time of MPs in the marine environment has significantly increased compared to the fate computed by Corella-Puertas et al. (2023). This is attributed to the addition of the sediment compartment in the present study. More specifically, while sedimentation was considered a net loss process in the previous study, in this work, we considered sedimentation as a process transferring MPs to the sediment compartment where the received mass can lead to additional exposure and effects on sediment-dwelling organisms. The other main fate mechanism in the water compartment, namely degradation, is calculated following the methodology of Corella-Puertas et al. (2023). Therefore, our residence time in water is in the same order of magnitude for all polymers previously considered.

Our simplified fate model does not consider aggregation and biofouling (Section 2.1), which are likely to increase the density and size of plastic particles. This, in turn, may increase the sedimentation rate of low- and medium-density polymers, hence potentially increasing their fate in the sediments while decreasing their fate in water (Hajjar et al., 2025). Future research should aim at quantifying these mechanisms in the marine environment.

4.2. Sediments exposure and effect factor

EEF_{sed} was found to be significantly smaller than EEF_w. We hypothesize that this is likely due to a difference in exposure estimates for MPs in water and sediments, rather than a difference in the sensitivity of water- or sediment-dwelling species. We investigated the sensitivity of aquatic organisms to pesticides in the Pesticide Properties Database, which contains effect data for various aquatic species, for both water and sediments (Lewis et al., 2016). Multiple entries in this pesticide database show that Lethal Concentrations of 50 % (LC50s) values of numerous pesticides are similar for water- and sediment-dwelling organisms (EC10 data is unavailable in the database). LC50s are a particular type of EC50s indicating mortality as the measured effect (usually single-dose studies, i.e. acute EC50) which can be used to derive chronic EC50eq and EC10eq for effect factors (EFs) calculations using extrapolation factors (Aurisano et al., 2019). For example, Atrazine has the same LC50 for *Chironomus riparius* (sediment-dwelling species) and

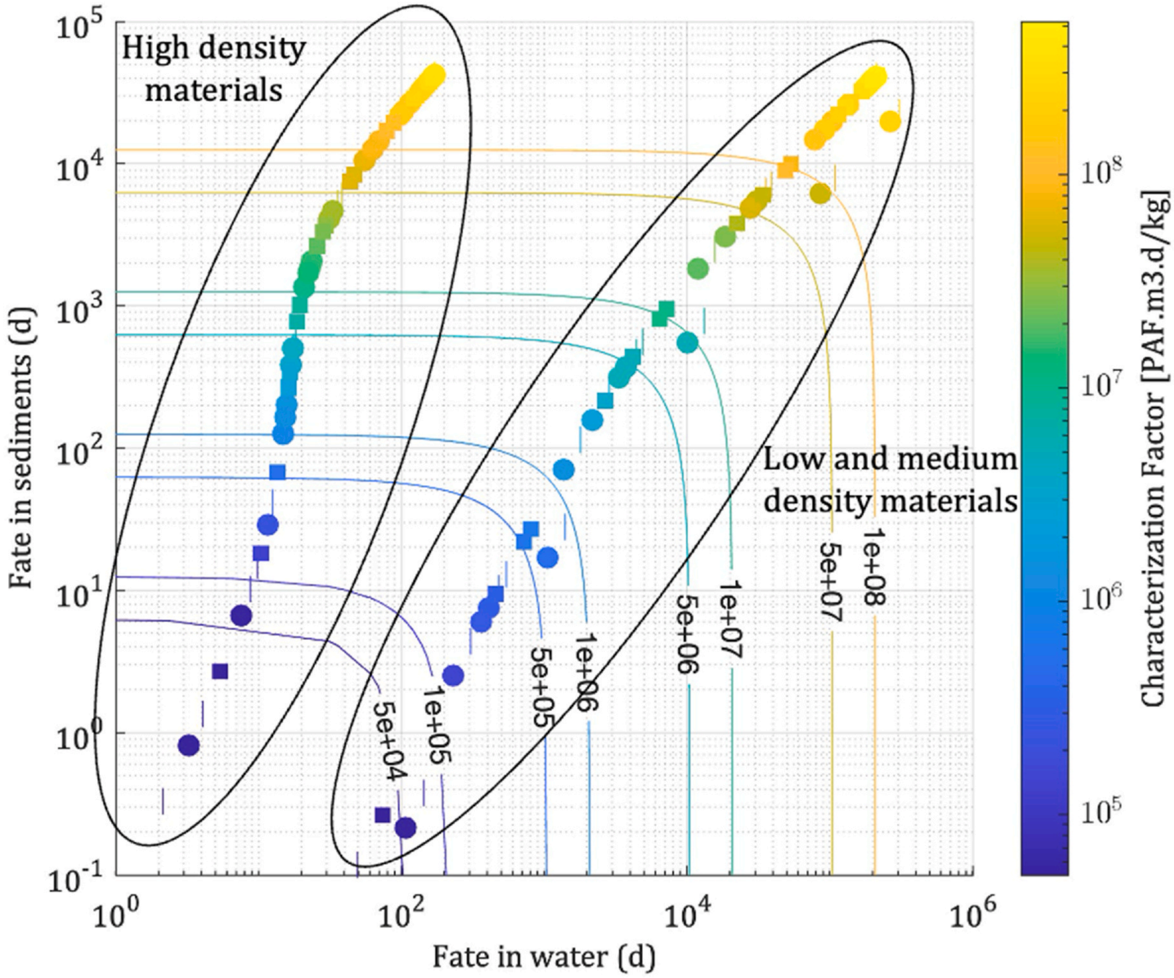

Fig. 4. Marine physical effects on biota midpoint CFs of films (squares), fibers (lines) and spheres (dots) microparticles, for an emission in marine water as a function of fate in water and sediments. The microplastic size (1, 10, 100, 100, 5000 μm) is represented by the marker size (e.g. smallest marker = 1 μm micro plastic). Underlying data can be found in Appendix E.

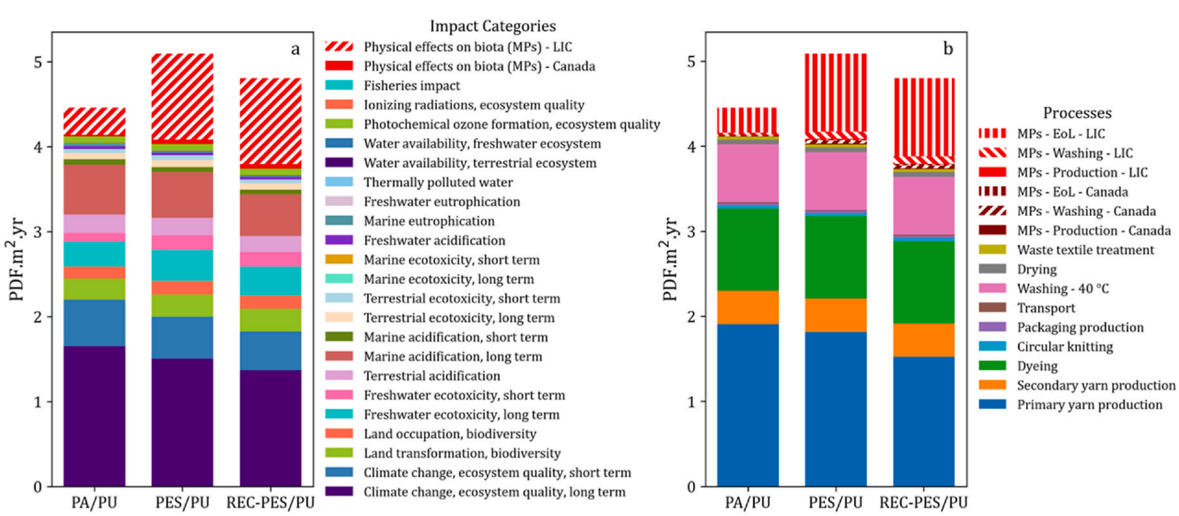

Fig. 5. Impacts on ecosystem quality of physical effects on biota of microplastics emitted over the life cycle of a pair of leggings in Québec, compared to other impact categories (a, left). Impacts on ecosystem quality per life cycle stage, for all impact categories, of a pair of leggings in Québec (b, right). Impacts of microfibers released are shown in red for a use phase in Canada and in a low-income country (LIC). (For interpretation of the references to color in this figure legend, the reader is referred to the Web version of this article.)

Table 2
Default physical effects on biota characterization factors (CFs) of aquatic microplastic emissions of different polymer densities (low, medium, high) and shapes (abbreviated in the table: microbeads/spheres/fragments of unspecified shape, microfibers/cylinders, microplastic film fragments). More specific CFs for different polymers, shapes and sizes can be found in Appendix E. PVC: Polyvinyl Chloride, PAN: Polyacrylonitrile, PS: Polystyrene, PLA: Polylactic Acid, HDPE: High-density polyethylene, LDPE: Low-density polyethylene, TRWP: Tire and road wear particles, PU: Polyurethane, PET: Polyethylene terephthalate, EPS: Expanded polystyrene, PP: Polypropylene, PA: Polyamide, PHA: Polyhydroxyalkanoate.

Shape	Polymer type	Default size	Midpoint CFs (PAF m^3 day/kg$_{emitted}$)		Endpoint CFs (PDF m^2 year/kg$_{emitted}$)	
			Marine	Freshwater	Marine	Freshwater
Microplastic beads/spheres	Low-density polymer <0.8 g/cm^3	1000 µm	5.17E+07	3.88E+07	1.41E+03	1.06E+03
	Medium-density polymer 0.8–1.1 g/cm^3		4.13E+08	3.10E+08	1.13E+04	8.48E+03
	High-density polymer >1.1 g/cm^3		3.25E+08	3.25E+07	8.90E+03	8.90E+02
Plastic microfibers/cylinders	Low-density polymer <0.8 g/cm3	10 µm	6.81E+05	5.11E+05	1.86E+01	1.40E+01
	Medium-density polymer 0.8–1.1 g/cm3		2.73E+07	2.05E+07	7.48E+02	5.61E+02
	High-density polymer >1.1 g/cm3		4.64E+07	4.64E+06	1.27E+03	1.27E+02
Microplastic film fragments	Medium-density polymer 0.8–1.1 g/cm3	100 µm	2.27E+08	1.70E+08	6.20E+03	4.65E+03
	High-density polymer >1.1 g/cm3		2.22E+08	2.22E+07	6.08E+03	6.08E+02
Microplastic beads/spheres	EPS	1000 µm	5.17E+07	3.88E+07	1.41E+03	1.06E+03
	HDPE		2.42E+08	1.82E+08	6.63E+03	4.97E+03
	LDPE		1.63E+08	1.22E+08	4.46E+03	3.35E+03
	PA/Nylon		7.55E+07	7.55E+06	2.07E+03	2.07E+02
	PET		2.15E+08	2.15E+07	5.88E+03	5.88E+02
	PHA		3.38E+06	3.38E+05	9.24E+01	9.24E+00
	PLA		3.25E+08	3.25E+07	8.90E+03	8.90E+02
	PP		5.81E+07	4.36E+07	1.59E+03	1.19E+03
	PS		4.13E+08	3.10E+08	1.13E+04	8.48E+03
	PVC		3.77E+08	3.77E+07	1.03E+04	1.03E+03
	TRWP		2.51E+08	2.51E+07	6.87E+03	6.87E+02
	PU/Spandex		2.39E+08	2.39E+07	2.52E+03	6.53E+02
	PAN/Acrylic		3.78E+08	3.78E+07	4.23E+02	1.03E+03
Microfibers	EPS	10 µm	6.81E+05	5.11E+05	1.86E+01	1.40E+01
	HDPE		6.73E+06	5.05E+06	1.84E+02	1.38E+02
	LDPE		3.29E+06	2.46E+06	8.99E+01	6.74E+01
	PA/Nylon		1.40E+06	1.40E+05	3.83E+01	3.83E+00
	PET		1.34E+07	1.34E+06	3.67E+02	3.67E+01
	PHA		1.47E+04	1.47E+03	4.02E-01	4.02E-02
	PLA		4.64E+07	4.64E+06	1.27E+03	1.27E+02
	PP		3.69E+05	2.77E+05	1.01E+01	7.58E+00
	PS		2.73E+07	2.05E+07	7.48E+02	5.61E+02
	PVC		2.04E+08	2.04E+07	5.59E+03	5.59E+02
	TRWP		2.00E+07	2.00E+06	5.47E+02	5.47E+01
	PU/Spandex		1.67E+07	1.67E+06	4.58E+02	4.58E+01
	PAN/Acrylic		2.03E+08	2.03E+07	5.56E+03	5.56E+02
	Cotton		7.16E+04	7.16E+03	1.96E+00	1.96E-01
	Viscose		1.04E+05	1.04E+04	2.83E+00	2.83E-01
	Wool		1.63E+05	1.63E+04	4.46E+00	4.46E-01
	Linen		3.79E+04	3.79E+03	1.04E+00	1.69E-02
Microplastic film fragments	HDPE	100 µm	8.84E+07	6.63E+07	2.42E+03	1.81E+03
	LDPE		5.03E+07	3.77E+07	1.38E+03	1.03E+03
	PA/Nylon		2.10E+07	2.10E+06	5.75E+02	5.75E+01
	PET		1.04E+08	1.04E+07	2.85E+03	2.85E+02
	PHA		5.80E+05	5.80E+04	1.59E+01	1.59E+00
	PLA		2.22E+08	2.22E+07	6.08E+03	6.08E+02
	PP		1.10E+07	8.23E+06	3.00E+02	2.25E+02
	PS		2.27E+08	1.70E+08	6.20E+03	4.65E+03
	PAN/Acrylic		3.58E+08	3.58E+07	9.79E+03	9.79E+02
	PU/Spandex		1.22E+08	1.22E+07	3.34E+03	3.34E+02
	Acrylic		3.58E+08	3.58E+07	9.80E+03	9.80E+02

Americamysis bahia (water-dwelling species). Multiple entries show similar results, such as Bensulfuron-methyl, Carbosulfan, Fludioxoni, i.e. a very close LC50 for a sediment-dwelling organism and a water-dwelling. These values, while not used in the proposed EEF, offer some insight: similar LC50s in water and sediments would mean that the EFs (computed from EC10eq extrapolated from LC50s) of these pesticides are also similar for sediments and water, meaning that water- and sediment-dwelling species have similar sensitivity to these chemicals. Following a similar trend, it can be hypothesized that the EF of MPs is also similar for sediments and water. In other words, EF_{sed} is as a first proxy assumed to be equal to EF_w until better data become available. This would mean that the different exposure factors in each compartment (XF_w and XF_{sed}) drive the difference in orders of magnitude between EEF_{sed} and EEF_w as EEF is the product of XF and EF.

Höss et al. (2022) argue that the higher food density in the sediments explains the higher ECx values in this compartment compared to the water column. In fact, organisms living in the sediments are less exposed to MPs as these organisms do not need to go through a large volume of sediments to recover the amount of food needed for their survival (small XF_{sed}). On the other hand, food is limited in the water column, as it is much less concentrated in organic matter and nutrients. Thus, organisms need to go through a larger amount of water to find the amount of food necessary, hence increasing the exposure to MPs (large XF_w) and consequently the EEF_w value compared to EEF_{sed}.

4.3. Characterization factors

Derived marine CFs range from 34 to 5.38×10^8 PAF m^3 d/ kg at midpoint level, and from 9.30×10^{-3} to 1.41×10^4 PDF m^2 yr/ kg endpoint level. The uncertainty of the CFs, calculated using Monte Carlo simulations as 95 % confidence intervals (CI) was determined to span approximately one order of magnitude above and below the CF values, for all materials (Appendix C). The simulation was done assuming a log-normal distribution for all the variables of the FFs and EEFs, namely, the degradation, sedimentation, deep burial and resuspension rates, as well as $HC20_{EC10}$, water depth and severity factor (Appendix C).

Our CFs of large, high-density microparticles are up to four orders of magnitude higher than the CFs computed by Corella-Puertas et al. (2023) as they did not include impacts in sediments. The size of high-density materials greatly influences their CFs, which was not the case when the sediments compartment was not included. In fact, high-density materials sediment quickly, and barely degrade while in the water column, making size the main factor influencing fate in the sediments, once the particles reach that compartment. Large particles indeed take longer time to degrade than small ones. Fig. 3 shows that midpoint CFs, for the same material, are three to four orders of magnitude higher for 5000 μm compared to 1 μm. Therefore, our conclusions are different from the ones of Corella-Puertas et al. (2023) since low and medium-density polymers show physical impact on biota of the same orders of magnitude as high-density polymers. Indeed, their results showed much lower CFs for high-density particles (Fig. 3).

Schwarz et al. (2024) computed CFs for MPs (for PP, LDPE and PET) in the marine environment. However, the CFs developed in our work differ in methodology. Specifically, their EF is based on an $HC50_{EC50}$ which does not meet the updated recommendation for an EF based on an $HC20_{EC10}$ (Owsianiak et al., 2023), and their CFs do not consider sediments. Our marine CFs are between one and four orders of magnitude larger than theirs, where the largest difference is seen for the PET (a high-density polymer), which sediments quickly and thus is considered to reach a compartment where MPs have no potential impacts in their model.

As a sanity check, we compared the contribution of per-capita MP emissions to the global average damage on ecosystem quality per capita per year, provided by Bulle et al. (2019) (9×10^4; PDF m^2 yr/ capita/ yr). It is estimated that 200 g of primary MPs are released into marine ecosystems per person per year (European Environment Agency, 2022). These MPs have an average size of 20 μm and are mostly irregularly- or fiber-shaped (Kooi and Koelmans, 2019). Using this information, we calculated an average impact of 155 $(8.35 - 2.87 \times 10^3)$ PDF m^2 yr/capita/yr for primary MPs, corresponding to 0.17 (0.01 − 3.19) % of the global average damage on ecosystem quality per capita, per year. This number shows a small contribution of primary MPs to overall damage on ecosystem quality. However, it neglects secondary MPs emissions (resulting from fragmentation of leaked macroplastics to MPs) which are more abundant than primary MPs in the marine environment (Duis and Coors, 2016).

4.4. Species distribution factor and scaling of compartmental impacts

It is worth noting that the CFs developed in this work are not directly comparable to other CFs at midpoint level (expressed in PAF m^3 d/ kg) as our CFs consider the total fraction of affected marine species, over a volume of marine compartment, rather than compartmental impacts traditionally computed in LCIA. Indeed, other LCIA impact categories do not compute CFs representing ecosystems as a whole as, until now, they only considered one compartment per ecosystem (e.g. water in freshwater ecotoxicity). To be comparable, the same compartments need to be included, while also using the SDF approach.

The SDF approach proposed in this work was used for the first time to derive ecosystem-level CFs. This allowed to include impacts of more than one compartments per ecosystem, here the marine water and sediments in LCIA. Further, this approach can be used in other ecosystems, for example in the freshwater ecosystem which also comprises different compartments, namely the rivers and lakes, each with their sediments and water compartments. The SDF matrix for this ecosystem could be developed using databases that contain data on the compartment in which freshwater species live. Similarly, the terrestrial ecosystem can be divided into natural soils and agricultural soils, and the SDF matrix can be obtained using the relative species richness of these two terrestrial compartments using species richness data gathered for the land use impact category (de Baan et al., 2013).

4.5. Case study

The results suggest that MPs could be a significant contributor to ecosystem quality damage. Depending on the location, leakage rate or end-of-life scenario and could affect the conclusions of an LCA. For example, in the LIC context of our case study, the ranking of the three different garments changes, as the Polyamide/Polyurethane (PA/PU) fabric becomes the best environmentally performing textile, due to the higher leakage rate in LIC and smaller CF of PA compared to Polyester (PES). Nonetheless, the impacts on ecosystem quality of the three pairs of leggings are mainly due to the climate change impact category. Emissions of greenhouse gases (GHG) mostly occur during the polymer production stages, as well as the dyeing and washing stages.

Thus, efforts should be made to reduce the GHG emissions over the life cycle of textiles, and more specifically during the production phase, or to increase the lifetime and number of wear of textiles. This study also shows that biodegradable fabrics, which aim at reducing the residence time of MPs in the marine environment, could have an effect on the overall LCIA of textiles. This eco-innovation could decrease impacts of leaked microfibers. Yet, this innovation could also introduce burden shifting, e.g. if the biodegradability agent is a highly toxic chemical, and should thus be further investigated.

While the marine ecotoxicity impacts are smaller than the impacts of MPs emitted on EQ (Fig. 5), the ecotoxicity of additives contained in MPs is not considered here, as it is still not well understood and requires further investigation. Including the impacts of these additives could lead to significantly higher marine ecotoxicity. Regardless, we expect ecotoxicity impacts in general to increase as this impact category transitions to effect factors based on $HC20_{EC10eq}$, which is not yet considered in Impact World + v2.1. In fact, Owsianiak et al. (2023) have shown that effect factors based on $HC20_{EC10}$ are on average 6.4 times higher than the ones based on $HC50_{EC50}$. Therefore, ecotoxicity impacts (marine, freshwater and terrestrial) of chemical emissions over the lifecycle of the garments studied are likely underestimated in the case study.

Additional case studies should aim at including the impacts on marine ecosystems of MPs emitted over the lifecycle of products and services. In the case of textiles, for example, different materials could be considered, while examining the effect of the production or use location, the number of washes, and the end-of-life scenario. Further, the developed CFs could be applied to LCAs in other sectors, such as plastic packaging or personal care products.

5. Conclusion

This work builds on the methodology proposed by MarILCA which aims to consider the physical effects on biota of micro- and nanoplastic emissions, within LCA. The proposed methodology adds the marine sediments compartment to the fate, exposure and effect proposed by MarILCA that only considered the water compartment so far (Corella-Puertas et al., 2023; Lavoie et al., 2022). The CFs results show that the addition of potential impacts in the sediments greatly increases the CFs of high-density materials as these particles reach the sediments compartment quickly. The CFs were tested in a textile case study, which showed that MPs emissions could be a significant contributor to ecosystem quality damage, depending on the scenario considered, and could potentially affect the conclusion of an LCA study.

This work developed and proposed a novel approach, the species distribution factor, to develop characterization factors for ecosystems that comprise multiple compartments (here water and sediments). For the first time, this allowed to include impacts in sediments in LCIA. Further, this approach can be used in other ecosystems, e.g. in the freshwater ecosystem which is comprised of rivers and lakes.

Further research should investigate the potential impacts of MPs in soil and freshwater ecosystems, to be included in the LCA of MPs emitting products and services, as these ecosystems are currently ignored in impact assessment methods. Moreover, other physical effects on biota impact pathways are investigated within the MarILCA framework (e.g. entanglement and ingestion of macroplastics, etc.), which will allow for comparison between various impacts of marine litter. This work therefore allows to include the impacts of MPs emitted over the life cycle of products and services in a more comprehensive manner and represents a step towards a more thorough evaluation of the environmental impacts of plastics.

CRediT authorship contribution statement

Nadim Saadi: Writing – review & editing, Writing – original draft, Visualization, Validation, Methodology, Investigation, Formal analysis, Data curation, Conceptualization. **Jérôme Lavoie:** Methodology, Formal analysis, Data curation, Conceptualization. **Peter Fantke:** Writing – review & editing, Methodology, Conceptualization. **Paula Redondo-Hasselerharm:** Writing – review & editing, Methodology, Investigation, Data curation. **Anne-Marie Boulay:** Writing – review & editing, Validation, Supervision, Resources, Methodology, Funding acquisition, Conceptualization.

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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Appendices. Supplementary data

Supplementary data to this article can be found online at https://doi.org/10.1016/j.jclepro.2025.146037.

Data availability

Data available in article supporting information and in the following Zenodo repository: https://doi.org/10.5281/zenodo.15599023.

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