It is time to develop characterization factors for terrestrial plastic pollution impacts on ecosystems in life cycle impact assessment: a systematic review identifying knowledge gaps

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LCIA OF IMPACTS ON HUMAN HEALTH AND ECOSYSTEMS

It is time to develop characterization factors for terrestrial plastic pollution impacts on ecosystems in life cycle impact assessment – a systematic review identifying knowledge gaps

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Abstract

Purpose Life cycle assessment (LCA) can help evaluate the environmental impacts of processes and products over their life cycle. However, the LCA community largely agrees that current assessment methods need further development to consider plastic pollution-related impacts on ecosystems. The present review identifies the knowledge gaps that need to be filled to develop characterization factors (CFs) considering the fate, exposure, and effects of plastic pollution within different environmental compartments and implement them in life cycle impact assessment (LCIA).

Methods A systematic literature review was carried out in the databases Web of Science and Scopus regarding the evaluation of plastic pollution in LCIA, followed by snowball sampling. In total, 59 relevant documents were found. Approaches regarding the modelling of fate, exposure, effects, and overall impacts were extracted, summarized, and critically analyzed to present the status of knowledge and deduct knowledge gaps.

Results Fate of plastic emissions considers their redistribution between environmental compartments, fragmentation, and degradation. Several approaches have been applied to model the redistribution of macro- and microplastics in different environmental compartments, but fragmentation has not been sufficiently integrated. There is one approach we found in literature related to degradation which is widely used. Exposure and effects have been modelled for the pathways entanglement, uptake, and greenhouse gas emissions. Additionally, seabed smothering and the transport of invasive species via plastic debris have been identified as pathways but their corresponding effects have yet to be quantified. For the marine compartment, all existing knowledge has been applied to LCIA. On the contrary, for the freshwater and terrestrial compartments, knowledge from the field of risk assessment still needs to be integrated.

Conclusions Knowledge is accessible for all fate processes for macro- and microplastics and has mostly been incorporated and applied to LCIA. On the contrary, not all exposure pathways have been adequately addressed. Especially for the terrestrial environment, a suitable definition of sub-compartments, a proper analysis of exposure pathways, and the translation of existing effect knowledge into EFs are lacking.

Keywords Plastic pollution · Life cycle impact assessment · Characterization factor · Terrestrial · Marine · Fate factor · Exposure factor · Effect factor

1 Introduction

Despite recent efforts of politicians, companies, non-governmental organizations, and individuals to reduce plastic pollution in the natural environment, plastics continue to accumulate in several environmental compartments worldwide (Li et al. 2016, 2020; Bergmann et al. 2019) with negative impacts on biodiversity and ecosystem services (Steinmetz et al. 2016; Horton et al. 2017; Wong et al. 2020). Life cycle assessment (LCA) is commonly used to determine and compare potential environmental impacts of processes and

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products over their entire life cycle by means of characterization factors (CFs) (ISO 14040:2009/Amd 1:2020 2009, ISO 14044:2006/Amd 2:2020 2006). CFs are specific to an elementary flow from the technosphere to the biosphere, which can be a resource used (extracted from the biosphere to be used in the technosphere) or a substance emitted (from the technosphere to the biosphere) and indicate its contribution to a specific environmental problem expressed as an impact category (Woods et al. 2021). To assess impacts in LCA, four aspects need to be considered (Jolliet et al. 2003, Woods et al. 2021; Maga et al. 2022): (i) quantified emissions, such as greenhouse gases (GHG) or plastic pieces, (ii) the fate of the emission in the environment, including its redistribution between environmental (sub-)compartments and possibly geographic regions, fragmentation, and degradation, (iii) exposure pathways and probabilities of exposure of organisms to the emission, and (iv) the effects of such exposure on the well-being of the organism, population, and ecosystem. The first aspect is addressed when compiling the life cycle inventory (LCI) data that quantifies the emissions related to unit processes; the remaining aspects are considered by the life cycle impact assessment (LCIA).

The LCA community largely agrees that current LCIA methods need further development to consider plastic-emission-related environmental impacts (Sonnemann and Valdivia 2017, Akdogan and Guven 2019, Schwarz et al. 2024), which are currently not considered in the most frequently used impact assessment methods (such as the Product Environmental Footprint (European Commission 2022), ReCiPe (Huijbregts et al. 2017), or IMPACT World + (Bulle et al. 2019)). Several research groups have addressed parts of the impact assessment method (e.g., Rosenbaum et al. 2008, Henderson et al. 2011, Fantke et al. 2018, Boulay et al. 2021, Woods et al. 2021). Nevertheless, a systematic methodology for assessing plastic pollution impacts on ecosystems in LCIA based on fate (FFs), exposure (XFs), and effect (EFs) factors is still lacking (Akdogan and Guven 2019, Allen et al. 2022, Sabate and Kendall 2024).

The present review aims to identify the knowledge gaps currently hindering the development of CFs considering fate, exposure, and effects of plastic pollution on ecosystems in all environmental compartments. The sub-objectives of this review are to (i) highlight existing knowledge and models regarding FFs, XFs, and EFs that can be used in LCIA to assess the impacts of plastic emissions on ecosystems and (ii) deduct knowledge gaps that currently hinder the development of suitable CFs and further research needs to provide a more complete assessment of the environmental impacts of plastic emissions. Hence, the current review focuses on the impact assessment phase within LCA rather than inventory compilation. It can be assumed that plastic pollution also has impacts on other areas of protection, such as human health, resource availability, or the economic, cultural, or natural value of structures (e.g., Beaumont et al. 2019, Yose et al. 2023) but these are out of scope of the current research. Our review contributes to the growing body of knowledge by addressing the research sub-objectives and adds to two recently published works, namely Sabate and Kendall (2024) who focused on LCAs of plastics related to their production and end-of-life but did not discuss plastic leakage in detail, and Xayachak et al. (2024) who focused on human health impacts and summarized polymer types, sizes, sources, and exposure pathways.

2 Methods

To identify existing knowledge regarding the integration of plastic pollution impacts in LCIA, a systematic literature review was carried out following the PRISMA 2020 requirements (Page et al. 2021). The databases Web-of-Science and Scopus were searched in early November 2024 using the following search terms in the article titles, abstracts, and keywords:

LCA OR “life cycle”

AND	*plastic OR *plastics OR polymer
AND	“characterization factor*” OR “characterisation factor*” OR “characterization model*” OR “characterisation model*” OR “fate factor*” OR “fate model*” OR “effect factor*” OR “effect model*” OR "exposure factor*"
AND Source type	Research article OR review
AND Language	English

Book chapters, conference proceedings, and short communications were excluded since they have not been subjected to peer review. Likewise, documents in languages other than English were excluded. All resulting documents were read by one researcher and classified as relevant or irrelevant according to their content. Documents that do not present fate, exposure, or effect factors or models for plastic pollution were excluded from further analysis. Likewise, articles describing the results of an experiment investigating a single effect or mechanism were excluded. No further exclusion criteria were applied. The search process is visualized in Fig. 1. By explicitly stating the search term and exclusion criteria in this review article, the authors make the search reproducible, reducing the risk of bias in the phrasing of the search term and the determination of the relevance of the documents.

From the identified documents, information regarding the modelling of fate, exposure, effects, and overall impacts was extracted, summarized, and analyzed to present the status of knowledge and deduct knowledge gaps. Afterwards, a snowball sampling approach was followed to include studies regarding the identified knowledge gaps that are not directly related to LCIA but contain information that can be adapted to LCIA purposes. That means documents cited in the identified documents were also screened and included if they provide information that can be used to deduct CFs for plastic emissions. The snowball sampling was continued until the saturation point, where adding more documents no longer provided new insights. It became clear that many existing models are designed for risk assessment rather than specifically for use in LCIA. That is the reason why many documents were identified during the snowball approach (37 documents) compared to the initial search with the term "LCA" or "life cycle" (22 documents).

3 Results

The search rendered 41 peer-reviewed research and review articles in the initial search and 63 during the snowball sampling (see Fig. 1). After applying the exclusion criteria, the resulting 59 documents were summarized and critically analyzed. The metadata of the relevant documents is presented in Table S1 in the Supplementary Material (e.g., year of publication, environmental compartments and size of emission considered (macro- or micro-/nanoplastics), etc.). Literature typically distinguishes macroplastics with at least one dimension greater than 5 mm and microplastics with all dimensions ≤ 5 mm. The latter is sometimes further divided into micro- and nanoplastics (all dimensions ≤ 1 µm). The review includes information for all three size classes.

More than half of the relevant documents were published in the last three years (since 2022) and only 20% are older than five years. Most documents were published

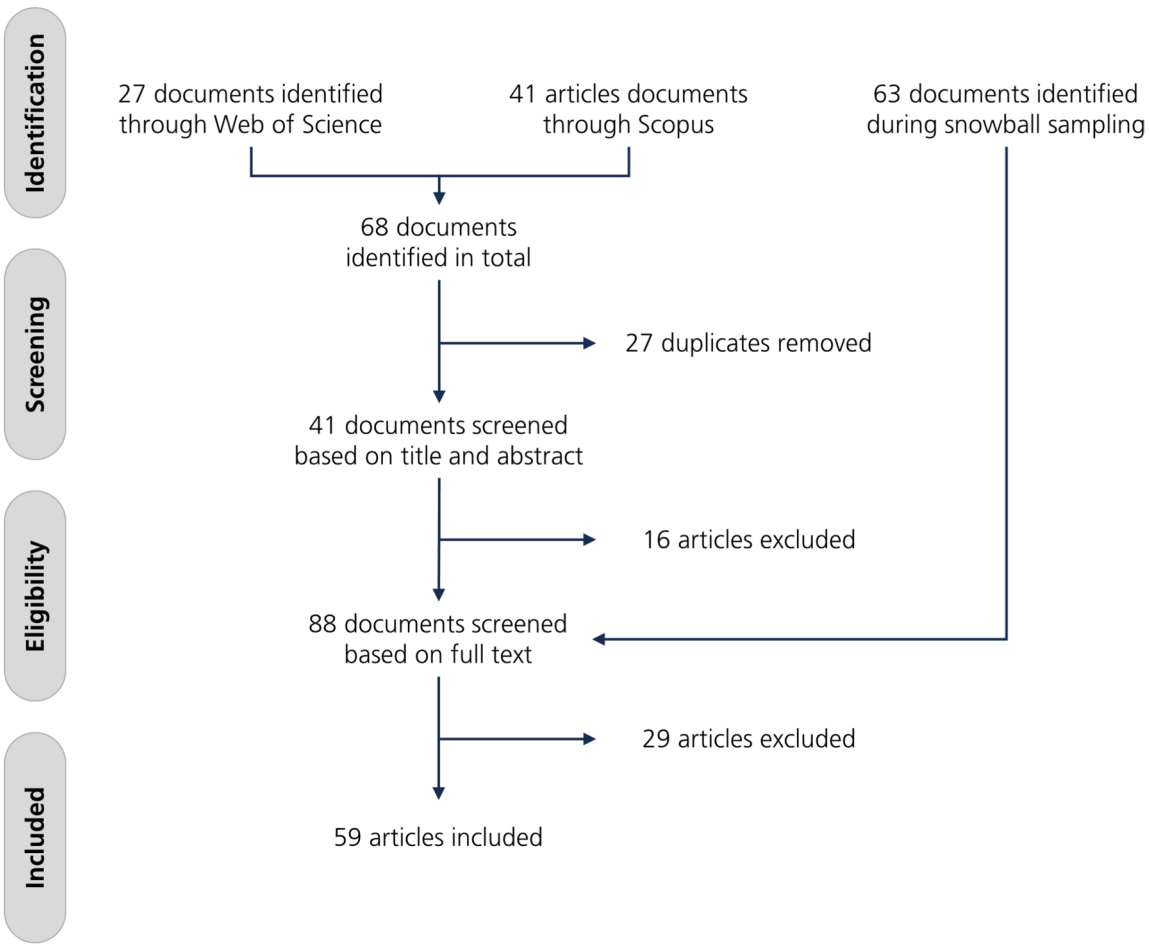

Fig. 1 Literature screening and inclusion process

in the International Journal of Life Cycle Assessment and the journal Science of the Total Environment (8 documents each), followed by the journal Environmental Pollution (4 documents). 32 documents describe (parts of) fate models and 34 address exposure and/or effects. 14 documents apply to macroplastics only, 27 to microplastics only, and 18 apply to both size classes or do not specify the size class. No document specifically targets nanoplastics. Instead, they are either implicitly included or explicitly excluded in the models. 28 documents include an LCA or aim at an integration of knowledge into LC(I)A. We drafted a framework for the characterization of plastic impacts in LCIA (see Fig. 2) based on Saling et al. (2020), Woods et al. (2021), and Piao et al. (2024), which we used to structure the existing knowledge regarding plastic pollution impacts on the environment (see Fig. 2).

Plastics of different sizes are emitted (also referred to as 'released') to different environmental compartments, where they are redistributed (transported or transferred within and between environmental (sub-)compartments), fragment (break down into smaller pieces), and degrade (polymer chains break, molecular mass decreases, and monomers mineralize (Andrady 2011, Gewert et al. 2015, Pauli et al. 2017)). Degradation causes GHG emissions (CO_2, CH_4, and non-methane volatile organic compounds (NMVOC)), contributing to global warming and photochemical ozone creation. Organisms get exposed to the plastic pieces that are not (yet) degraded via the following pathways as displayed in Fig. 2: entanglement, uptake (ingestion, adsorption and absorption), rafting, and smothering. As a result, organisms suffer toxic or physical effects or indirect plastic pollution induced impacts, e.g., global warming, photochemical ozone creation, or acidification, ultimately reducing species diversity, leading to deteriorated structure and functioning of the ecosystem. Physical effects on biota occur internally within the digestive tract and externally by entanglement, smothering, or adsorption to the outer membranes (e.g., epidermis). Smothering refers to the inhibition of gas exchange on the seabed by larger plastic items, leading to anoxic or hypoxic conditions (Woods et al. 2016 and 2021). The cause-effect pathways displayed in Fig. 2 are based on the aquatic environment. For the terrestrial environment, no cause-effect chain has been suggested yet. As a starting point, we assume that all aquatic pathways are generally also possible in the terrestrial environment. Nevertheless, it is necessary to validate this assumption and determine whether additional pathways exist for the terrestrial environment, to be comprehensive. For example, entanglement may be less prominent in the terrestrial environment, but the uptake pathway of microplastics may need to be extended to cover also plants, not just animals (Shafea et al. 2023).

While information is available for most fate processes, no documents presented information regarding the exposure pathways and corresponding effects of emissions of dissolved organic carbon from plastic emissions, rafting, and smothering. We synthesized all results and depicted the status of knowledge regarding (3A) fate and (3B) exposure pathways and corresponding effects and its integration into LCIA in Fig. 3. A limited application in LCIA refers to knowledge applied superficially, e.g., by using factors developed for a different compartment or factors based on expert opinion.

3.1 Fate modelling

The fate of plastic emissions in the environment considers (i) their redistribution between environmental (sub-) compartments, (ii) their fragmentation, and (iii) their degradation in these compartments. Woods et al. (2021) and Maga et al. (2022) considered the initial environmental compartments (the environmental compartment where a plastic item is first emitted): air, soil/terrestrial, freshwater, and marine

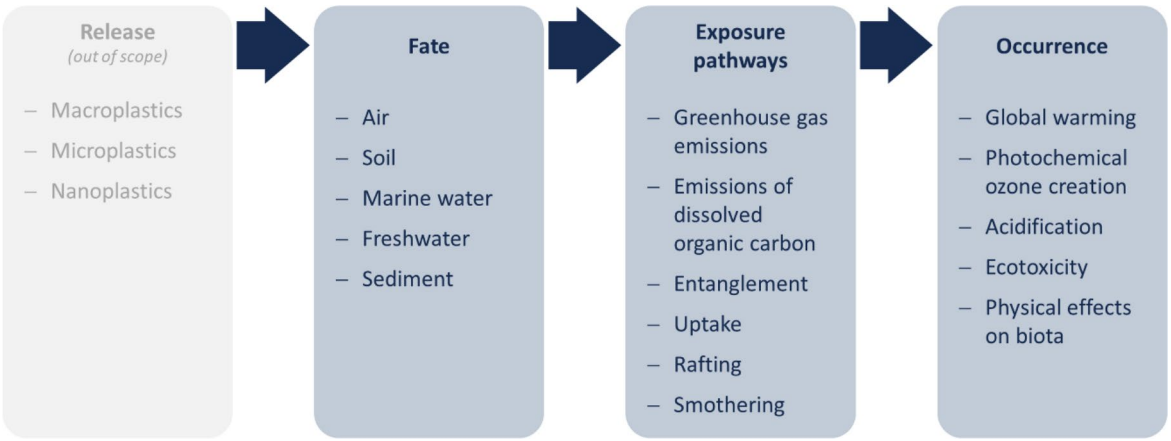

Fig. 2 Framework for the assessment of plastic pollution impacts on ecosystem quality

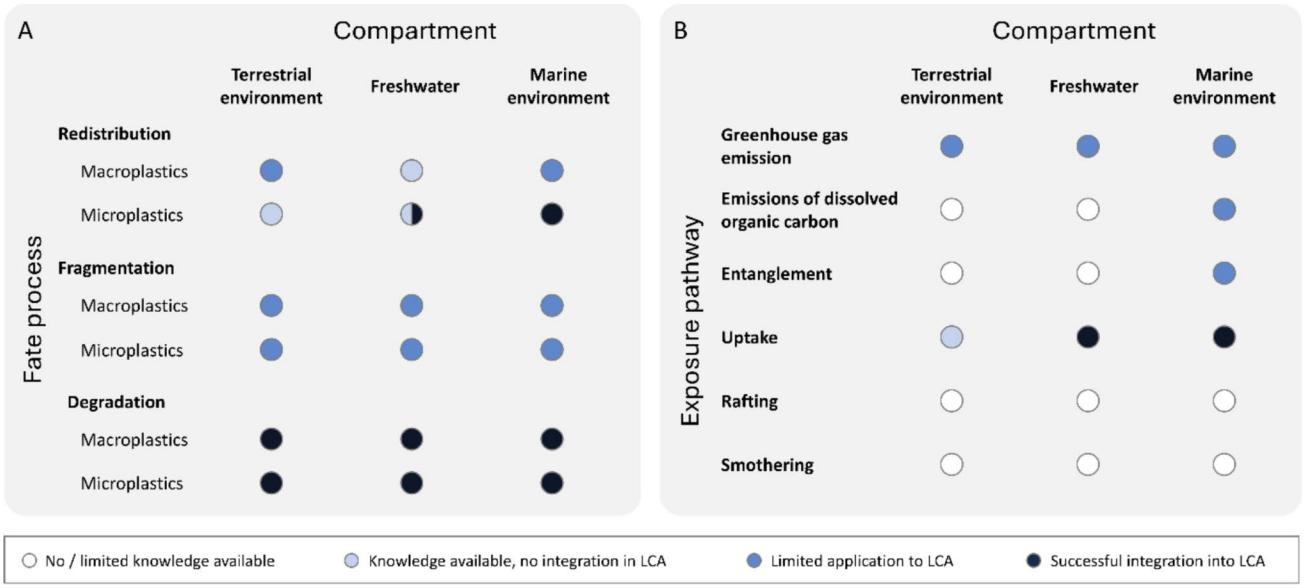

Fig. 3 Status of knowledge integration into LCIA regarding (A) fate and (B) exposure pathways for three different environmental compartments

water. Maga et al. (2022) defined the final compartments (the compartments where emissions accumulate and persist) as marine water, marine and river sediment, and soil/terrestrial, although the provided FFs do not differ between marine and river sediment due to a lack of data. Researchers collaborating as the MarILCA working group address the final compartment of marine water only. Contrary to Maga et al. (2022), Corella Puertas et al. (2022 and 2023) and Piao et al. (2024) considered sedimentation a removal mechanism; therefore, the sediment was not considered a compartment. Schwarz et al. (2019) and Hajjar et al. (2023) further detailed the resolution of the aquatic compartment, including distribution and removal mechanisms in several sub-compartments at the continental and global scale. Likewise, the SimpleBox4Plastics (SB4P) model for microplastics by Quik et al. (2023) considers two scales (regional and continental) with seven compartments each (air, lake water, freshwater, seawater, natural soil, agricultural soil, and other soil) and three zones (moderate, arctic, and tropical) with three compartments each (air, water, and soil). The SB4P model was not initially aimed at an integration into LCIA but has been adopted by other researchers, e.g., Schwarz et al. (2024).

3.1.1 Redistribution

Plastic redistribution has been modelled in terrestrial and aquatic environments with an emphasis on the aquatic compartment (22 of 28 documents vs. 13 addressing redistribution on soil) and microplastics (21 documents vs. 13 addressing macroplastics). Most redistribution models are specific to either macro- or microplastics. However, there is currently no fate model regarding the redistribution of plastic chemical additives.

Terrestrial environment Three approaches can be found in the literature to model the redistribution of macroplastic emissions to the terrestrial environment: one based on the emission's proximity to the ocean, one based on the emission's weight, and one based on the topography at the point of emission. Jambeck et al. (2015) assumed that 15—40% of mismanaged plastic waste generated within a 50 km distance from a coast is transferred to the ocean. For each country, the value within this range is an expert judgment considering local weather conditions, topography, vegetation, and removal infrastructure. While the initial model did not target an application to LCIA, several authors have used the redistribution rates in fate models for both macro- and microplastics (Croxatto Vega et al. 2021; Woods et al. 2021; Maga et al. 2022; Stafford et al. 2022; Tang et al. 2022; Galafton et al. 2023), despite the simplistic approach and the lack of scientific proof for the range or the influencing factors. Similarly, the approach by Civancik-Uslu et al. (2019) is a simplification: it assigns a higher impact score to a higher risk of redistribution, even though there is no scientific evidence that redistribution itself is harmful. More scientific models consider the topography at the point of emission by dividing the analyzed geographic area into a grid of evenly sized cells and determining a redistribution probability and direction for each cell. The redistribution probability is based on (sub-) surface runoff (Lebreton et al. 2017; Meijer et al. 2021; Mellink et al. 2022 and 2024), remobilization probability of deposited plastics during floods (Lebreton et al. 2017), and wind speed (Meijer et al. 2021; Mellink et al. 2022

and 2024) compared to corresponding thresholds, which depend on the land use and terrain slope. The redistribution direction depends on the average wind direction for wind transport and the distance-weighted difference in elevation above sea level for water transport. Terrestrial redistribution models assume that hydrological processes further redistribute plastics transported to rivers and that the emissions do not re-enter the terrestrial compartment. Although Lebreton et al. (2017) only consider positively buoyant plastics, their approach has been used as part of LCIA methodology by Woods et al. (2021), Tang et al. (2022), and Hajjar et al. (2023). Contrarily, the more sophisticated models of Meijer et al. (2021) and Mellink et al. (2022 and 2024) have not yet been applied to LCIA. Despite being quite complex and time-consuming to apply, the latter models appear to depict the redistribution of macroplastics from soil to freshwater most accurately. Their temporal and spatial resolutions can be adapted to match corresponding fate, exposure, and effect models. A limiting factor is that small-scale obstacles (e.g., hedges where plastics can get stuck, creating hotspots) are not considered. Simplifying this approach, Croxatto Vega et al. (2021) modeled the redistribution of microplastics from soil to the air depending on the wind speed at a certain location using a national average for different land use categories and the emission's size (which is, in turn, dependent on the degradation speed). Based on these results, we conclude that only values based on expert opinion (Jambeck et al. 2015; Civancik-Uslu et al. 2019) have been integrated into LCIA methodology for macroplastics emitted to soil.

A model of the redistribution of microplastics emitted to soil is the SB4P model (Quik et al. 2023), a multimedia mass balance modeling system in which the plastic masses in the environmental compartments are the steady-state solutions of mass balance equations for all compartments. The model considers the redistribution mechanisms (i) transport between compartments, such as surface runoff and erosion of soil grain, (ii) removal by transport outside the system, (iii) hetero-aggregation with colloidal natural particles or attachment to larger particles, and (iv) removal by degradation. The model has been applied to LCIA by Schwarz et al. (2024). None of the terrestrial-focused models suggests a vertical redistribution of macro- or microplastics into deeper soil layers or a transport by groundwater, although experiments by Mintenig et al. (2019), Panno et al. (2019), Weber and Opp (2020), and Chia et al. (2021) showed microplastic occurrence in groundwater, requiring such downward transport.

Freshwater Newbould et al. (2021) and Nakayama and Osako (2023) modeled the redistribution of macroplastics by rivers. The former focused on macroplastic trapping along meander bends, in overhanging vegetation, or along channel banks and divided the stream into a set of equally sized cells with different probabilities of trapping based on riverbed properties. The latter considered advection, dispersion, diffusion, and settling. Neither has been integrated into LCIA methodology.

For the redistribution of microplastics from freshwater systems (especially rivers) to the marine environment, several models describe both horizontal and vertical transport, as well as homo-/hetero-aggregation and sedimentation (burial) (e.g., Nizzetto 2016, Besseling et al. 2017; Zhao and You 2022; Domercq et al. 2022; Nakayama and Osako 2023). However, these models demand highly accurate hydrological and/or emission-specific input data, resulting in limited applicability (Mennekes and Nowack 2023). Instead, Mennekes and Nowack (2023) suggested a more generic hydrological model for larger scales, which has not yet been applied to LCIA. In the same year, Quik et al. (2023) presented the SB4P model, which has been applied to LCIA by Schwarz et al. (2024). Piao et al. (2024) also focused on microplastics in freshwater but only considered sedimentation and resuspension. Zhao and You (2022) were the only ones addressing redistribution from river water to the air by bursting bubbles. Nevertheless, depending on the surface area of the geographic region considered, atmospheric deposition may be neglected (Mennekes and Nowack 2023). Synthesizing these results, the redistribution of microplastics in freshwater shows two outcomes: one part highlights the need to evaluate the suitability of the model of Mennekes and Nowack (2023) for integration into LCIA, while the other part reflects the successful integration of the SB4P model.

Marine water In the marine compartment, for macroplastics, Maga et al. (2022) focused on the emission's buoyancy based on its density only and disregarded other redistribution mechanisms without an explanation. Høiberg et al. (2024) model geographic redistribution using a Lagrangian transport model based on particle positions recorded for a total of five years. The models for microplastics distinguish vertical transport (sinking, re-surfacing, and sedimentation; Hajjar et al. 2023; Quik et al. 2023) and horizontal transport by waves and wind (Hajjar et al. 2023; Schwarz et al. 2019), as well as hetero-aggregation/attachment, and sediment burial, and have all been integrated into LCIA.

3.1.2 Fragmentation

Fragmentation refers to the breaking of plastic emissions into smaller pieces without mineralization. Fragmentation, therefore, increases the number of pieces in the environment but not their total mass. Fragmentation has been modelled by Koelmans et al. (2017), Saling et al. (2020), Kaandorp et al. (2021), and Hajjar et al. (2023)

for the marine compartment, Nakayama et al. (2023) for freshwater, and Croxatto Vega et al. (2021) for emissions to soil. The SB4P model (Quik et al. 2023) applies to all environmental compartments. Koelmans et al. (2017) assumed that all macro- and microplastic emitted to the ocean fragments over time (including degradation, abrasion, and fragmentation) with mass-based rates proportional to the emission's surface area. The model of Croxatto Vega et al. (2021) includes a so-called degradation module, which is, in fact, a fragmentation module because it considers photo-degradation (UV damage to polymer surfaces by polymer type and plastic product usage resulting in surface-driven fragmentation) as well as oxidation. In the model of Saling et al. (2020), fragmentation occurs first, breaking the microplastic emission down into particles of 100 µm, at which point in time degradation starts without splitting the particles any further, thus maintaining the same number of fragmented particles until they are entirely mineralized. The fragmentation rate is based on weight loss data retrieved from literature (Artham et al. 2009; Sudhakar et al. 2007), neglecting the possibility that (some of) the weight loss might be due to degradation instead of fragmentation. For the degradation rate, no specific source is cited. The resulting FF indicates the number of fragmented particles at a point in time depending on the time horizon, regardless of the particle's size (≤ 100 µm), shape (spheres, irregular fragments, or fibers), or mass. The SB4P model (Quik et al. 2023) suggests default fragmentation rates found by Koelmans et al. (2017) and Kaandorp et al. (2021). The researchers collaborating as the MarILCA working group consider fragmentation as a mechanism that alters an emission's size class (e.g., removes microplastics (1 µm – 5 mm) and at the same time forms nanoplastics < 1 µm) (Hajjar et al. 2023). Woods et al. (2021) point out that the fragmentability, e.g. the longevity of a plastic emission in a certain size class, influences an emission's transport and possibly impacts. Redondo-Hasselerharm et al. (2024) stress this by pointing out the different size thresholds for ingestion and tissue translocation. Fragmentability depends on the shape and polymer type of the emission. For example, thinner emissions are more likely to fragment than thicker ones due to their reduced mechanical robustness (Hajjar et al. 2023). Different polymer types also influence the susceptibility to fragmentation (Song et al. 2017 and 2020). Fragmentation might change an emission's shape from film or fiber to particle (Alkema et al. 2022), which would influence the degradation speed used in the models of Maga et al. (2022) and Corella Puertas et al. (2023). However, both author groups neglect fragmentation. As a result, none of the existing fate models considers the effects of fragmentation on redistribution and degradation.

3.1.3 Degradation

All existing models assume a surface-driven degradation. That means the degradation time in a specific compartment depends on the emission's polymer type, size, and shape. Regarding the shape, the emission can be characterized as 1-dimensional (films), 2-dimensional (fibers), or 3-dimensional (nearly spherical pellets or particles) according to the object's smallest dimension, which dominates its fate. Corella Puertas et al. (2023) included a surface area correction factor accounting for the difference in surface roughness and porosity of real-life microparticle emissions compared to the idealized model of a sphere. The size refers to the emission's smallest dimension (a film's thickness or a fiber or particle's diameter, respectively). Degradation rates for the most widely used commodity polymer types have been summarized by Chamas et al. (2020) based on different degradation experiments in the compartments landfill/compost/soil (buried), water (with exposure to light), biological (in a laboratory with enzymes or microbes), and sunlight (comparable to the soil surface), deducing specific surface degradation rates (SSDRs) in µm/year, distinguishing between test samples with and without fillers and additives. They also calculated the expected half-lives of the polymer types. This method was also applied and extended by Salieri et al. (2021), Maga et al. (2022), Corella Puertas et al. (2022 and 2023), Galafton et al. (2023), Quik et al. (2023), Schwarz et al. (2024), and Piao et al. (2024) but none of these research groups searched degradation rates systematically. Besides, some researchers included experiments that test the degradability according to certain test protocols (e.g., in industrial compost) but not the actual degradation under (near-)natural conditions. Synthesizing the information, the approach to determine surface-driven degradation rates developed by Chamas et al. (2020) is well integrated within LCIA for micro- and macroplastics within all compartments.

Additionally to excluding values from experiments under non-natural conditions, Maga et al. (2022) apply a data quality assessment to assess data uncertainty and only consider the dataset(s) with the lowest coefficient of variation per polymer and compartment. The data quality assessment considers the dataset's reliability, completeness (alignment of the experiment to real-world conditions and degree of degradation measured), temporal and geographic correlation, and applied measurement method. Contrarily, Corella Puertas et al. (2022 and 2023) worked with three different degradation rates that are primarily based on conservative assumptions regarding the slowest degradation rate for the slow scenario (0.001 µm $year^{-1}$ for most polymer types) and the highest value of the literature review for the fast scenario. For the medium degradation scenario per polymer type, they use the geometric mean of the degradation rates found in literature, without

considering uncertainty. This approach neglects differences in dataset quality and aspects such as the geographic representativeness of a dataset. For example, Hajjar et al. (2023) point out the effect of UV intensity, water temperature, and oxygen diffusion on degradation rates, which vary geographically and seasonally. Stafford et al. (2022) developed a simpler model that calculates the persistence of an emission in the environment based on the biodegradability (not actual biodegradation) according to standards for industrial composting, seawater, and anaerobic sludge at temperatures that are not realistically reached in the environment.

3.2 Modelling exposure and effects

For CFs to link a certain plastic emission to environmental impacts, they need to consider the exposure of organisms to the emission and the corresponding effects. The exposure refers to the share of plastic in an environmental (sub-)compartment that is bioavailable to cause effects in organisms and varies with the emission's shape, size, concentration, and the organism's behavior (e.g., playful animals are more likely to get entangled in lost/abandoned fishing gear) (Lawson et al. 2015). Besides, organisms can be exposed to plastics via the dietary route, specifically by consuming other organisms that have previously ingested plastic (Zhu et al. 2019). The corresponding XF varies between 0 and 1. Assessing the exposure to plastic emissions is challenging because they are diverse in sizes, shapes, and composition (polymer types and additives), they change properties over time due to fragmentation and degradation, and their abundance in nature is extremely variable on spatial scales (Koelmans et al. 2017 and 2022). Likewise, the bioavailability of plastic additives may change over time, fostered by ageing and fragmentation (e.g., Artham et al. 2009) and ingestion leading to exposure of the plastic to gastrointestinal gut fluids (e.g., Koelmans et al. 2013). Koelmans et al. (2014) developed a corresponding model to quantify the uptake of additives from ingested plastic by marine organisms but found that the uptake of bisphenol A and nonylphenol in lugworm and cod via plastic ingestion was negligible.

The EF addresses the adverse effects caused by the exposure of organisms to plastic emissions, such as mortality or reduced reproductivity. We discuss XFs and EFs together since they are often combined into joint exposure-effect factors (XEFs) (e.g., Lavoie et al. 2021; Høiberg et al. 2024). In these cases, the XF is set to 1 (all emitted plastic is directly bioavailable). Only for additives, XFs still need to be developed, e.g., based on Bridson et al. (2023).

3.2.1 Gas emissions during degradation

Croxatto Vega et al. (2021) linked plastic pollution impacts to global warming via the emission of GHG during mineralization. They applied gas production rates calculated by Royer et al. (2018), who conducted experiments to quantify the production of hydrocarbon gases from emitted plastics under natural conditions. Similarly, Zhao and You (2022) estimated GHG emissions from degradation based on the assumed polymer type per microplastic shape (fibers: 100% Polyethylene Terephthalate (PET), fragments: 34% PET and 66% Polyethylene (PE), foam: 100% Polystyrene), corresponding hydrocarbon gas emission rates, and corresponding CFs taken from the ecoinvent V3.8 Database. Likewise, Piao et al. (2024) linked microplastics in sediment and sinking from water to the sediment to global warming caused by the GHG emissions resulting from anaerobic and aerobic biodegradation, respectively. The quantities of emitted GHG were derived based on the theoretical amount of CO_2 that would be produced during the complete degradation of the emission based on its carbon content and the SSDRs based on Chamas et al. (2020), adapted by Maga et al. (2022) and Corella-Puertas et al. (2023). The authors pointed out the trade-off between the reduced persistence of biodegradable plastics in the environment on the one hand and GHG production during their degradation on the other hand, which stresses the impact of weighting different impact categories. As for global warming, Croxatto Vega et al. (2021) and Zhao and You (2022) estimated NMVOC emissions from degradation and applied CFs of existing LCIA methodologies to estimate potential photochemical ozone creation impacts (van den Oever et al. 2024).

3.2.2 Entanglement

For the marine compartment, Woods et al. (2019) quantified the effects of chronic entanglement in macroplastics based on the share of species per group that inhabits a certain geographic region and has been reported entangled (e.g. Gall and Thompson 2015). The effects were expressed in terms of a potentially affected fraction of species (PAF) per unit of marine floating plastic density (g km^{-2}) and are spatially and species-specific. This unit is gaining more attention (see Fantke et al. 2018); however, cannot directly be linked to mass-based LCI data. Note that the model covers floating marine debris only, neglecting the effects of submerged plastic objects. Høiberg et al. (2022 and 2024) refined the entanglement factors of Woods et al. (2019) by better matching the spatial dispersion of plastic debris and species distributions. Contrary to the fate models described above, the presented XEFs for entanglement are not specific to the polymer type, size (any diameter larger than 4.75 mm), and shape of the emission, as data regarding the total current

pressure was taken from Eriksen et al. (2014), who provided data for all polymer types, sizes, and shapes combined. However, it may be questioned whether entanglement is entirely independent of the size and shape of floating marine litter (Woods et al. 2021) and the type of product such as fishing gear, and it is definitely species-dependent.

3.2.3 Uptake

The uptake of plastics by animals can lead to physical impacts such as intestinal tract obstruction and, consequently, starvation, growth, hormone production, fitness, behavior changes, reproduction, and more (Gall and Thompson 2015). Similarly, smaller microplastics can also be taken up by plants (Li et al. 2020) and affect plant responses, e.g., inhibit growth and nutrient uptake by physical blockage. Besides, the uptake of plastics can cause ecotoxicity effects. Over 1300 chemicals of concern are known to be marketed for use in plastics and 29 – 66% of the chemicals used or found in well-studied plastic types are of concern (Wagner et al. 2024). Therefore, physical effects can only be distinguished from ecotoxic effects if experiments are conducted with non-additivized polymer types and additives separately. In practice and unlike the researchers collaborating as the MarILCA working group, we assume that a variety of mixtures affects the organisms and it is a joint impact by the particle fractions and the chemicals shedded from the particles (Vijver 2019). The uptake of macroplastics has not been discussed in literature because these fragments are most likely too large to be absorbed. Thus, all available knowledge refers to microplastics.

EFs for the exposure pathway via ingestion by animals or uptake by plants are typically calculated based on a comparison of exposure concentrations and threshold effect concentrations (Koelmans et al. 2020) using species sensitivity distributions (SSDs). SSDs can be based on acute or chronic physical toxicity data from laboratory tests performed on single species (Lavoie et al. 2021; Loubet et al. 2022; Li et al. 2023). Such tests may result in acute or chronic EC_x values (median effect concentration; the concentration at which x % of the effect occurs), LC_x values (lethal concentration; the concentration at which x % of the exposed population dies), LOEC values (lowest observed effect concentration; the lowest tested concentration with effects that differ statistically significantly from the control), or NOEC values (no observed effect concentration; the tested concentration immediately below the LOEC without a statistically significant effect compared to the control). Conversion factors are applied to convert acute into chronic values. According to the USEtox model, which is typically used in LCIA to characterize aquatic toxicity effects, EFs need to be based on data from at least three different species covering at least three trophic levels to be considered representative of the whole ecosystem (Rosenbaum et al. 2008). XEFs are given in potentially affected species (PAF) $m^3\ kg^{-1}$. Lavoie et al. (2021), Salieri et al. (2021), Tang et al. (2022), and Casagrande et al. (2024a) calculated EFs using the hazardous concentration HC_{50} (concentration at which 50% of the tested species are potentially affected), obtained as the geometric mean of EC_{50} values ($HC_{50}EC_{50}$). For future calculations, Owsianiak et al. (2023) recommend using the HC at the 20th percentile using chronic EC_{10}-equivalents to resemble concentrations found in the environment more closely.

The approaches for calculating (X)EFs vary in (i) the toxicological dose descriptors studied, (ii) the endpoints considered, and (iii) the compartments considered. Regarding the dose descriptors, 15 out of 17 studies used NOEC values, 12 studies used EC_{50} values and LOEC values, respectively, and 9 used LC_{50} values. 5 studies consider all four dose descriptors. If the shape of the dose–response curve is known, all toxicological dose descriptors can be recalculated from each other. The disparity in available dose descriptors influences the results: In the study of Lavoie et al. (2021), the exclusion of LOEC and NOEC values led to a change of the EF from 82.28 PAF $m^3\ kg^{-1}$ to 72.9 PAF $m^3\ kg^{-1}$. Regarding the endpoints considered, the most frequent one is reproduction, followed by mortality and growth (see Table S2). Regarding the compartment in focus, some studies focus on a single compartment, e.g., marine (2 studies), freshwater (2 studies) or soil (5 studies). However, 8 studies present XEFs based on a combination of marine and freshwater species to meet the USEtox requirement for sufficient data. Uptake effects of microplastics have been quantified by Gall and Thompson (2015) and Everaert et al. (2018) for marine water, and Adam et al. (2019) for freshwater. Their findings have been translated into XEFs for physical effects on biota by Lavoie et al. (2021) for the aquatic compartment in general and Salieri et al. (2021) for freshwater. Likewise, Tang et al. (2022), Casagrande et al. (2024a), and Holmquist et al. (2018) converted data regarding impacts caused by the uptake of plastic additives in the aquatic environment into EFs for ecotoxicity. According to Lavoie et al. (2021), uptake effects are independent of the emission's shape and size (within the range of microplastics). Terrestrial uptake effects have been qualitatively described by Shafea et al. (2023) and quantified by Jacques and Prosser (2021), Wang et al. (2022), Tunali et al. (2023), and Redondo-Hasselerharm et al. (2024). However, this information has not yet been translated into XEFs for the terrestrial environment.

Based on the shape and size of microplastics used in toxicity experiments, EFs for the uptake pathway might be underestimated. Redondo-Hasselerharm et al. (2023) and Casagrande et al. (2024b) analyzed the match between the physical properties of plastics used in ecotoxicity experiments and to calculate EFs on the one hand and plastics ingested by organisms on the other hand. Casagrande et al.

(2024b) observed that experiments were more frequently conducted with regularly shaped plastics (beads or pellets, together 72%) rather than fibers and fragments, representing 63% and 22% of shapes found in the natural environment, respectively. They referred to studies indicating that irregularly shaped plastics take longer to be egested as they adhere to tissue surfaces (Choi et al. 2018; Kolandhasamy et al. 2018; Gonçalves et al. 2019) and may, therefore, cause more damage than regularly shaped microplastics. Besides, Casagrande et al. (2024b) pointed out that the share of polymer types used in experiments differs from the one of production and, thus, emission. Likewise, Koelmans et al. (2020) pointed to a nonalignment of microplastic particles used in effect tests regarding their sizes, shapes, and polymer types. To solve this, they provided a method to correct for such differences. Synthesizing the results, we conclude that generic EFs that apply to several different polymer types only accurately reflect actual effects if the polymer type and shape composition of the experimental data resembles the one found in the environment.

De Ruijter et al. (2020) and Redondo-Hasselerharm et al. (2023) recommend applying a quality assurance and alignment to experimental data used to determine uptake effects. Redondo-Hasselerharm et al. (2023) first assigned data quality scores of 0, 1, or 2 to 20 criteria in 4 categories: particle characterization, experimental design, applicability for risk assessment, and ecological relevance. For the SSD, they only chose datasets with a summed score of at least 20 out of 40 and non-zero values for 5 particular criteria. As a second step, they aligned these datasets by rescaling their size range and effective concentration to be more environmentally relevant and consider bioaccessibility under real-world conditions.

3.3 Characterizing impacts

Impacts can be assessed at midpoint (problem) or endpoint (damage) levels (Huijbregts et al. 2017). While midpoint indicators address immediate environmental effects and are more accessible to calculate due to their proximity to the impact source, endpoint indicators provide a holistic view of the long-term consequences that is more relevant for decision- and policymaking and communication to non-LCA practitioners. It is still unclear whether plastic pollution impacts require the definition of a new (midpoint) impact category or whether they can entirely be integrated into existing impact categories. Several authors integrate plastic pollution into existing categories. Other researchers define a new midpoint impact category called 'plastic litter' or 'plastic pollution'. For example, Saling et al. (2020) presented a CF at the midpoint level for the marine compartment in the unit of pellet equivalent points compared to the reference unit of PE beads. The CF depends on the number and shape of fragments created during a specific time period. In general, the CF is lower for emissions that fragment more slowly and whose fragments degrade relatively fast, keeping the concentration of fragments low. The authors used this CF as a starting point for further optimization and do not outline a possible conversion to endpoint impacts or an integration with other approaches, such as the USEtox-based approach to integration uptake effects.

Two other approaches used normalization factors to set plastic pollution impacts in relation to other impacts. Zanghelini et al. (2020) and Galafton et al. (2023) normalized plastic pollution impacts based on an average global plastic pollution and applied different weighting factors. Contrarily, in the methodology applied by Civancik-Uslu et al. (2019), Stefanini et al. (2021), and Gao and Wan (2022), the resulting scores are divided by the maximum value of the investigated alternatives. These approaches can, therefore, only be applied when comparing different alternatives and do not allow a comparison across studies.

The integration of aquatic plastic pollution impacts into an existing impact assessment methodology was demonstrated by Corella Puertas et al. (2022). They combined their fate model with the XEF of Lavoie et al. (2021) after converting this XEF according to the recommendation of Owsianiak et al. (2023). Corella Puertas et al. (2022) expressed impact scores as midpoint CFs in the unit CTUe $kg_{emitted}^{-1}$ (comparative toxic unit for aquatic ecotoxicity impacts according to the USEtox model (Fantke et al. 2017)), which is equivalent to PAF m^3 d kg^{-1}, and as endpoint CFs following the methodology used in ImpactWorld + (Bulle et al. 2019). Likewise, Salieri et al. (2021) applied their newly developed XEF based on Adam et al. (2019) to FFs based on degradation rates extracted from Chamas et al. (2020). Alternatively, all USEtox-based (X)EFs can be converted into endpoint effects expressed as the potentially disappeared fraction of species (PDF) m^3 d kg^{-1} by applying a severity factor (Jolliet et al. 2003; Bulle et al. 2019; Owsianiak et al. 2023), which was done by Schwarz et al. (2024) and Piao et al. (2024).

4 Discussion

When conducting the systematic literature search using the search term listed above, including "life cycle" or "LCA", only 23 of 59 relevant papers were identified (see Fig. 1). The remaining documents were found during the snowball sampling, making it evident that a lot of relevant literature is not targeted at the application in LCIA but at other areas, such as risk assessment. Some of the pertaining information collected by environmental scientists and ecotoxicologists has been applied to LCIA methodologies by integrating it into FFs, XFs, or EFs. For example, for

the marine environment, all documents not directly targeting LCIA applications have been incorporated into other studies to inform LCIA methodologies, leaving no relevant knowledge from other fields unutilized in LCIA. This means that redistribution rates or ecotoxic effects have either been incorporated into methodologies through their application in LCA studies or deemed unsuitable with a justified explanation. On the contrary, documents were available for freshwater and the terrestrial environment that address the fate, exposure, or effects of macro- and microplastics but have not yet been applied to LCIA (see Table S1 in the Supplementary Information). An obstacle to the integration of risk assessment data in LCIA lies in the different focuses of the data gathering: while toxicity experiments usually apply a receptor perspective, LCA follows an emitter perspective (Askham et al. 2023). It is, therefore, necessary to translate or convert the available information for use in LCIA. It should be noted that impacts on human health, e.g., particulate matter formation via the inhalation of air-borne plastics by humans (Croxatto Vega et al. 2021), as well as other risks that are atypical for LCIA, such as increased flood risk caused by drains clogged by plastic litter (Njeru 2006; van Emmerik and Schwarz 2020), are outside the scope of this review.

While it seems that the redistribution of both macro- and microplastics can be modelled in all relevant compartments based on existing knowledge, note that the models' resolutions and definitions of the compartments vary. Although higher resolutions of the compartments provide more detail (e.g., dividing the marine compartment into benthic, pelagic, shoreline, etc.), they also reduce practicability for LCA practitioners. Besides, the sensitivity of LCA results to the choice of resolution has not yet been investigated. Additionally, while sub-compartments have been suggested for the soil on different scales based on the land use (natural soil, agricultural soil, urban soil (Quik et al. 2023)), there is no scientific evidence that this distinction resembles the drivers of fate, exposure, and effect mechanisms in the terrestrial compartment, which may instead lie in, e.g., the flora, fauna, climate zone (temperatures, rainfall), surrounding (e.g., windbreakers/corridors), etc. Similarly, the existing models apply the common distinction of macro- and microplastics but do not specifically address nanoplastics. It has also not been investigated whether the size limits used to distinguish macro- from micro- and nanoplastics are indeed the most appropriate ones for LCIA. For example, the probability of tissue translocation by plastics is significantly higher for plastics sized < 83 µm (Redondo-Hasselerharm et al. 2023), which does not correspond to the common size categories. Once all relevant exposure pathways are sufficiently understood, the LCA community can evaluate whether it would be useful to deviate from the common size categories to better address the resulting effects.

4.1 Characterization of aquatic plastic pollution impacts

The existing knowledge regarding the marine environment contains information regarding the redistribution, fragmentation, and degradation of both macro- and microplastics, as well as exposure, entanglement, and uptake (ingestion) effects (see Table S1 in the Supplementary Information). However, some of the EFs for the uptake pathway were developed for the aquatic environment in general (including marine and freshwater) due to a lack of data. As effects are usually studied on either marine or freshwater species, with sufficient data available, it would be recommended to develop separate EFs per compartment.

The aquatic fate models neglect geographic redistribution, which would be necessary to account for regional impacts, be compatible with entanglement effect models, and increase relevancy for policymakers (Høiberg et al. 2024). As a starting point, Hajjar et al. (2023) point out mechanisms governing the horizontal transport of microplastics in the marine compartment. The integration of geographic transfer into the redistribution model would enhance compatibility with spatially explicit EFs, such as the entanglement factors developed by Høiberg et al. (2024). Besides, the LCA community needs to make decisions regarding the treatment of sedimentation as a compartment or redistribution mechanism, including a distinction between sedimentation and deep burial.

For the aquatic environment, three aspects are unresolved: First, the XEF regarding the entanglement pathway needs to be further detailed to consider the size and shape of emissions, and the unit needs to be adapted to make it compatible with the results of the fate modelling. Regarding the integration with environmental impacts, it needs to be determined how the differentiation of effects on the five species groups can be considered. Second, regarding the uptake pathway, XFs addressing the migration of additives from plastic emissions over time depending on their degradation are lacking (e.g., based on Luo et al. (2022) and Markic et al. (2020)). Additionally, the trophic transfer of plastics is neglected within the EFs. Third, the impact category addressing the formation of biofilms, more specifically the transport of invasive species or other harmful pollutants on plastic emissions, has only partly been investigated (Ziccardi et al. 2016), and representative XEFs are lacking. Concluding, these knowledge gaps do not hinder the development of CFs and are already being addressed by various research groups. As case studies have shown, it is already possible to characterize aquatic plastic pollution impacts as part of an LCA (Corella-Puertas et al. 2023; Schwarz et al. 2024).

4.2 Characterization of terrestrial plastic pollution impacts

According to the Organisation for Economic Cooperation and Development, 90.7% of plastic in 2019 was emitted to soil, of which only 32% was redistributed to the aquatic environment (2022), leaving roughly two-thirds of all plastic emissions on soil. This finding is supported by material flow analysis, e.g., Schwarz et al. (2023). In 2016, Nizzetto et al. pointed out the lack of scientific attention paid to plastic emissions on soil. While current European Union projects place emphasis on this, such as the PAPILLONS project (European Union (Ed) n.d), data is still being created and has not yet been published nor curated in large amounts.

The models of Mellink et al. (2022) and Quik et al. (2023) can be adapted to address the redistribution of macro- and microplastics, respectively, from soil to other compartments, and degradation rates have been summarized by, e.g., Chamas et al. (2020) and Croxatto Vega et al. (2021). What is still needed is the integration of the macroplastic redistribution model (Mellink et al. 2022) to LCIA, as well as the development of XEFs for the terrestrial compartment. As our literature review showed, knowledge regarding the effects of terrestrial plastic pollution on different ecotoxicological endpoint indicators (e.g., mortality, reproduction, oxidative stress, morphotoxicity, cytogenotoxicity) for different target organisms (e.g., soil itself, plants, animals) is available. Nevertheless, this knowledge is not yet in a form that could be used for (X)EFs. Besides, the USEtox model used for the (X)EFs for the aquatic environment currently only applies to aquatic species, and common LCIA methods do not include an impact category accounting for terrestrial ecotoxicity. Therefore, even with structured information regarding EC_x or LC_x values of different polymer types, sizes, and/or shapes in the terrestrial environment, such as provided by Redondo-Hasselerharm et al. (2024), the translation of such data into XEFs is not as straightforward as the information regarding the aquatic environment. Additionally, the relevant impact pathways for the terrestrial environment are yet to be determined, e.g., entanglement might be less relevant here than in aquatic environments. It is, therefore, currently not possible to characterize possible terrestrial plastic pollution impacts in LCIA. Likewise, but with less pollution accumulating, exposure and impacts in sediments have not yet been comprehensively addressed. When developing XEFs for the terrestrial compartment, the similarity between sediments and soil may be used to justify applying findings from one to the other.

4.3 Implications for further research

Based on the distribution of emitted plastics, the most imminent implication for further research is to integrate all existing knowledge into a macroplastic fate model and to determine the most relevant impact pathways for the terrestrial environment. Based on these findings, XEFs should be developed for the most relevant impact pathways to enable a more comprehensive assessment of plastic pollution impacts in LCA, including the terrestrial compartment. Regarding the fate model, independent of the environmental compartment, the LCA community needs to make certain decisions regarding a suitable surface-area correction factor for real-life emissions compared to idealized shapes, the regarded time horizon, as well as a uniform suggestion of size classes and values for polymer types with unknown degradation speed. Besides, more research is necessary to incorporate fragmentation into the fate model. The inclusion of fragmentation models may result in the adoption of an amount-based approach (number of particles) in exposure and effect modelling rather than or additionally to the currently predominant mass-based approach, as applied, e.g., in the risk assessment approach of Jacques and Prosser (2021).

While assessing data quality and uncertainty in LCA is more common regarding LCI data, it would also be relevant for the elements of the characterization factor, which are based on experiments under varying conditions. For example, degradation tests may be conducted in the laboratory only mimicking natural conditions and isolating certain mechanisms. In cases where data is limited, using all available data is necessary to fulfill requirements such as posed by the USEtox methodology. Where data is more abundant, data may be chosen based on a quality assessment (Maga et al. 2022; Redondo-Hasselerharm et al. 2023) or results may indicate the full range between the minimum and maximum value (Corella Puertas et al. 2022). Both approaches lead to limitations regarding the consistency within the method as certain CFs would entail a higher uncertainty than others based on data availability. Curating sufficient data and applying consistent data quality assessment is necessary to enable comprehensive uncertainty analysis in LCA, not only focusing on LCI data.

From the larger perspective, the question arises, which is the most appropriate way to set the resulting impact scores of plastic pollution in relation to other impact categories and, thus, provide a larger context. Previous case studies have presented impact scores at the midpoint (Saling et al. 2020; Zanghelini et al. 2020; Galafton et al. 2023; Civancik-Uslu et al. 2019; Stefanini et al. 2021; Gao and Wan 2022) and endpoint level (Corella-Puertas et al. 2023; Schwarz et al. 2024). Nevertheless, the former requires a normalization and highly subjective weighting of impact categories to obtain information at a level suitable for decision-making. For the latter, the existing methodology cannot account for terrestrial impacts.

For all aspects of the impact assessment framework, there needs to be a balance between accurate representation of the physical, biological, and chemical processes in the environment on the one hand and the practicability of derived CFs

and the ease of comprehension and interpretation of the results on the other. For LCA practitioners, this is especially relevant during inventory modelling (e.g., regarding the exact composition of plastic emissions as additives are not always specified in detail, and their shape and size may be unknown) and interpretation of results, including sensitivity and uncertainty analysis and hotspot identification. This is crucial for stakeholders that make policies or decisions based on LCA outcomes when communicating the results and underlying assumptions and uncertainties and deriving solutions.

4.4 Limitations of the available models

All mentioned existing models and approaches entail limitations and uncertainties addressed in the respective documents. For example, literature-based degradation data used in the fate models stems from experiments with varying degrees of representation of natural conditions, varying accuracy and suitability of measurement methods, and varying experiment duration to account for seasonal or otherwise temporal changes in degradation speed. Besides, the authors of such experiments may or may not know and specify the exact composition of the plastic emission(s) investigated, including all additives, and the composition of the microorganisms available in the studied environment that might facilitate degradation. Similarly, regarding the effects of plastics and additives, experiments were conducted under different test conditions in different geographic regions and with organisms at different life stages, and conversion factors were applied in many cases to obtain EC_x values and to convert acute to chronic values. The selection of data points used to calculate the EFs can influence the EFs for some additives by one order of magnitude, as becomes apparent when comparing the results of Casagrande et al. (2024a) to those of Tang et al. (2022) and the USEtox database. Additionally, EFs for the uptake pathway are largely based on experiments measuring effects at the individual level (e.g., mortality, growth, reproduction) rather than at the population level (e.g., abundance) and almost never based on the community level (e.g., biodiversity, species composition).

Another limitation is the compatibility of LCI data, FFs, XFs, and EFs. Whereas LCI data is usually mass-based, FFs according to the methodology developed by Maga et al. (2022) are given as $kg_{PPe}\ kg_{emitted}^{-1}$ (PPe referring to plastic pollution equivalents), which can be interpreted, e.g., as the amount of time 1 kg of plastic persists in the environment (in years) or as the plastic mass that persists in the environment for one year (in kg). However, this unit is not typical in LCIA and integration with XFs and EFs is not intuitive. In the methodology of Corella Puertas et al. (2022 and 2023), FFs were presented as $kg_{compartment}$ day $kg_{emitted}^{-1}$ and the results of the SB4P model (Quik et al. 2023) for microplastics are given as the Predicted Environmental Concentration of microbeads in g/m^3 or particles/m^3. None of these is directly compatible with EFs for the entanglement pathway, which were expressed as PAF per unit of marine floating plastic density ($g\ km^{-2}$) (Woods et al. 2019; Høiberg et al. 2024). While workarounds can be found for modelling the foreground system in LCA, this incompatibility hinders the inclusion of plastic pollution impacts in LCA databases. As a result, case studies tend to underestimate plastic pollution impacts. The problem of the connectivity between LCI data and LCIA methods is further enhanced by the need to report certain impacts geospatially explicitly, which is usually not considered in available databases.

5 Conclusions

The status of knowledge availability and integration into LCIA varies for different environmental compartments, emission sizes, fate processes, and exposure pathways. Nanoplastics have not been explicitly addressed but are covered by many of the microplastic models. Nevertheless, it must be investigated whether the size limits used to distinguish macro- from micro- and nanoplastics are indeed the most appropriate ones for LCIA. Besides, since different impact pathways apply to different size classes, fragmentation needs to be better integrated into LCIA. Regarding redistribution, the status of knowledge integration into LCIA is slightly better for microplastics than for macroplastics. For the latter, models are available for the terrestrial environment and freshwater but have not yet been applied to LCIA. For the marine environment, horizontal redistribution has been modelled but the inclusion of vertical transfer is still very simplistic. For degradation, knowledge has been applied to LCIA for all compartments and size classes. The status is worse for exposure and effects: The pathways of emissions of dissolved organic carbon, rafting, and smothering have been identified, but not assessed sufficiently. Entanglement effects have been quantified for the marine environment but not for freshwater and the terrestrial environment. Besides, an integration with existing fate models has yet to be developed. Uptake effects of microplastics (not additives) are the only ones that have been successfully integrated into LCIA for the aquatic environment. For the terrestrial environment, they have only been described and measured but not yet applied to LCIA. In general, the terrestrial environment lacks a suitable definition of sub-compartments, a proper analysis of exposure pathways, and the translation of existing effect knowledge into EFs. As a result, it is only possible to assess the fate and effects resulting from GHG emissions during degradation and microplastic uptake by organisms in the aquatic environment but not in the terrestrial environment.

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Declarations

Conflicts of interest The authors have no competing interests to declare that are relevant to the content of this article.

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