PHA from Wastewater: A Promising Strategy for Reducing Microplastic and Macroplastic Emissions

AUTHORS
VALENTINA H. PAUNA, SIMON ALEXANDER SAXEGÅRD & CECILIA ASKHAM

REPORT NO.
OR.03.23

YEAR
2023

ISBN NO. 978-82-7520-908-3
ISSN NO. 2703-8610
REPORT TYPE Report
CONFIDENTIALITY Open

PROJECT NUMBER
3383

PROJECT TITLE
Scoping Investigation of Microplastics and Macroplastics in the Context of LCA and Preliminary Exploration of Environmental Impacts

COMPANY CONTACT
Valentina H. Pauna

REQUESTER
Invest-NL

CONTACT PERSON (S)
Guy de Sevaux and Xandra Weinbeck

APPROVED
Irmeline de Sadeleer

NUMBER OF PAGES
41

KEYWORDS
Microplastics, Life Cycle Impact Assessment, PHA, Material Flow Analysis, Biodegradable Plastic

PHOTO FRONTPAGE
Taken in Naples, Italy (May 2019) by Valentina H. Pauna

Abbreviations

CF	Characterization Factor
EF	Effect Factor
EC50	Median Effective Concentration
EPS	Expanded Polystyrene
FF	Fate Factor
MarILCA	Marine Impacts in Life Cycle Assessment (Project)
HC50	Hazardous Concentration
HDPE	High density polyethylene
HV	Hydroxyvalerate
LC50	Lethal Concentration
LCA	Life Cycle Assessment
LCIA	Life Cycle Impact Assessment
LCI	Life Cycle Inventory
LDPE	Low density polyethylene
MarILCA	Marine Impacts in Life Cycle Assessment (Project)
MFA	Material flow analysis
MP	Microplastics
PA	Polyamide
PE	Polyethylene
PHA	Polyhydroxyalkanoates
PHB	Polyhydroxybutyrate
PLP	Plastic Leak Project
PP	Polypropylene
PU	Polyurethane
TRWP	Tire and Road Wear Particles
XF	Exposure Factor

Policy Brief

Researchers, the public, governing bodies, and industry increasingly express concerns related to the possible environmental and human health impacts that could be associated with microplastic (MP) and macroplastic leakage. While the area of study has made significant and rapid progress, there remains critical uncertainty in the actual assessment of MP impacts.

From the literature and material flow analysis (MFA) calculations we draw the following conclusions:

- Plastic releases to nature have a high risk of causing impacts on ecosystems and humans.
- Macroplastic emissions from mulch film, geotextiles and dolly ropes can be avoided using PHA products (no microplastics left after 6 years).
- The potential replacement of traditional fossil-based plastics with PHA is 100% for mulch film, 100% for control-release fertilizer, approximately 50% for geotextiles and 100% for dolly ropes.
- Microplastics from PHA are degraded by bacteria, fungi and other biological processes and thus the risk of damage to organisms over time is much lower than for traditional plastics.

Uncertainty stems from a variety of aspects, particularly related to the complex ways in which MPs are leaked from the manufacturing of plastic products, their use, and the way in which they are discarded at their end of life. In order to include the effects of micro- and macroplastic on humans and ecosystems in LCA, the impact assessment methods are under development. No LCA studies of plastics and their alternatives currently include all of the relevant fate and effect pathways for microplastics and macroplastics into the life cycle impact assessment models that they use to calculate damage potentials on ecosystems and human health. Until these impact assessment methods have been developed, it is not feasible to carry out a quantitative impact assessment for comparison.

Despite this, it is possible to estimate the potential for replacement of conventional plastic using PHA and the subsequent reduction of MP emissions due to the use of PHA in place of conventional plastic (Figure 11). The use of biodegradable materials could be especially effective in products with a function that is suitable for such materials and that are known to cause a significant release of macroplastics and MPs into the environment. Therefore, this study aimed to investigate mulch film, control-release fertilizer, geotextiles and dolly ropes. Figure 11 shows that PHA could be an effective replacement material in terms of reducing MP emissions to the environment, especially for mulch film, control-release fertilizer, and dolly ropes. This study focuses on macro- and microplastic emissions associated with the products studied and no other potential environmental impacts.

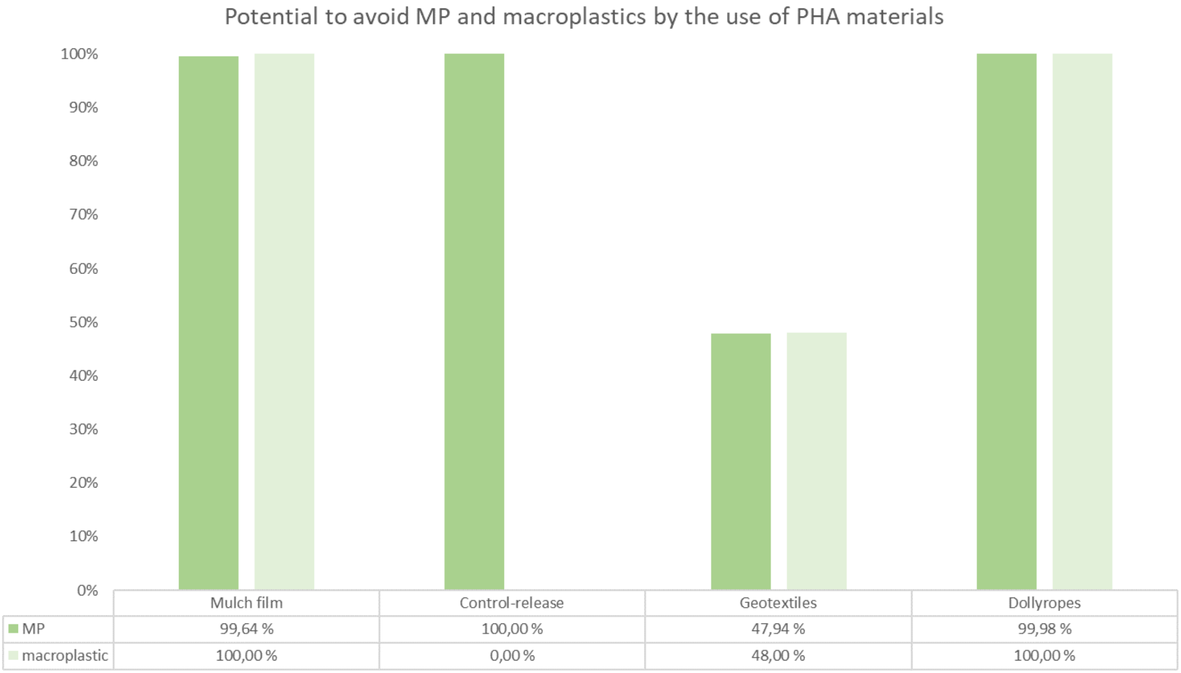


Figure 11 (Section 5.3). Relative avoided fossil plastic, MPs and microplastics leakage by using PHAs.

Technical Summary

Researchers, the public, governing bodies, and industry increasingly express concerns related to the possible environmental and human health impacts that could be associated with microplastic (MP) and macroplastic leakage. While the area of study has made significant and rapid progress, there remains critical uncertainty in the actual assessment of MP impacts. This uncertainty stems from a variety of aspects, particularly related to the complex ways in which MPs are leaked from the manufacturing of plastic products, their use, and the way in which they are discarded at their end of life.

Regardless of there not being an impact assessment to refer to, concerns related to MPs and macroplastics have already sparked interest in developing alternative biodegradable and bio-based materials to replace petroleum-based materials. One possible alternative involves the use of polyhydroxyalkanoates (PHA). There are several plastic products that would be good candidates for some or all of their composition to be made from PHA and for the purposes of this study, we have focused on mulch film, geotextiles, control-release fertilizer and dolly ropes.

To date, the literature has provided information on possible MP and macroplastic impacts and options to demonstrate these impacts in a quantitative manner. One such method that is under development, is Life Cycle Impact Assessment (LCIA). LCIA is an important phase of Life Cycle Assessment (LCA), an effective method for communicating possible environmental concerns with decision makers. Therefore, researchers have worked to develop each aspect of LCIA, many of which build upon one another, with the goal of establishing a method for LCA of MPs and macroplastic leakage.

While there is yet to be an agreed upon method for LCA of MPs and macroplastics, it is still possible to discuss and hypothesize possible impacts, particularly by way of estimating the emissions of MPs and macroplastics. Material flow analysis (MFA) is an effective method for tracing potential emissions through a product value chain. Therefore, this method has been applied in the present study, allowing us to investigate the possible contribution of selected products to MP and macroplastic leakage to the environment.

The MFA demonstrates that macroplastics have the largest leakage potential for most product groups. However, some product groups are more likely to directly degrade into MPs. The flow of MPs and macroplastics in the environment and its slow degradation rate means that species are exposed to the same plastics several times, both within the initial environmental compartment it was leaked to and in the following environmental compartments it travels to. The exact mass of plastic leakage that are caused by a product remains highly uncertain, however, it is clear from this study that the use of PHA could reduce macroplastic leakage and significantly mitigate MP leakage to the environment.

Contents

Abbreviations 3

Policy Brief 4

Technical Summary 6

1 Introduction 9

2 Background 9

2.1 Microplastic Overview 9

2.2 Polyhydroxyalkanoates 10

2.3 A Brief Overview of Life Cycle Assessment of Microplastics 11

2.4 Case Studies 12

2.4.1 Mulch Films 13

2.4.2 Geotextiles 13

2.4.3 Control-Release Fertilizer 14

2.4.4 Dolly Ropes 14

3 Keyword Co-Occurrence Analysis 15

3.1 Bibliometric Network Analysis 15

3.2 Method 15

3.3 Results 15

3.4 Discussion 18

4 Literature Review 19

4.1 Method 19

4.2 Results 19

4.3 Discussion 21

5 Material Flow Analysis 21

5.1 Background 21

5.2 Method 22

5.3 Results ... 26

5.4 Discussion ... 32

6 Simplified Impact Assessment ... 33

7 Conclusion ... 33

References ... 35

1 Introduction

This report is the final deliverable for NORSUS project no. 3383: *Scoping Investigation of Microplastics in the Context of LCA and Preliminary Exploration of Environmental Impacts*. Specifically, the work presented here intends to provide preliminary quantitative insight regarding the potential environmental impacts of MP emitted via use of petroleum-based plastics (petro-polymer(s)) for mulch film, geotextiles, control-release fertilizer, and dolly ropes. These four products were selected to be included in the following analysis because they have been described as good candidates (see Bauchmüller et al. 2021) for the use of biodegradable materials (specifically polyhydroxyalkanoates (PHA)) in place of their typical petro-polymer composition.

Invest-NL and the PHA2USE Consortium requested that NORSUS carry out a life cycle assessment (LCA) of relevant products to assess the impacts directly associated with the emission of microplastics (MPs). Understanding these impacts can help in determining the possible benefits of replacing some or all petro-polymer material used in MP emitting products with biodegradable materials. However, the inclusion of MPs as an impact category in LCA is an area of study currently under methodological development and NORSUS is involved in this progress through their participation in the scientific committee in the MarILCA (Marine Impacts in LCA) project (MarILCA 2022). Therefore, the present research project was defined as a scoping study which discusses the possible consequences of MP pollution and how this MP problem can potentially be weighed and compared to other environmental issues in the future, using LCA. It is important to note that the results presented here should not be interpreted as a finalized quantitative representation of MP impacts.

Given that the assessment of MPs in LCA, particularly Life Cycle Impact Assessment (LCIA), is under development, NORSUS has carried out several research activities to augment existing data in the literature and generate preliminary quantitative results to describe the possible impacts of MPs on the environment (described in Chapters 3 to 6).

Chapter 2 presents a background to the topic of interest. Chapter 3 provides an overview of the keyword co-occurrence analysis carried out using bibliometric network analysis followed by Chapter 4 which summarized a literature review focused on impact assessment of MPs. Next, Chapter 5 describes the material flow analysis (MFA) method used in this study to populate a life cycle inventory (LCI). Then, Chapter 6 discusses a simplified impact assessment. Chapters 3 to 6 first introduce the analytical methods used in this study, describe the methodological approach, present the results, and interpret the findings. Finally, Chapter 7 offers concluding remarks.

2 Background

2.1 Microplastic Overview

Microplastic (MP) pollution has been discussed in scientific literature since the 1970s by Carpenter and Smith (1972) and Carpenter et al. (1972). Since then, numerous authors from various fields of study have worked together to define MPs (GESAMP, 2016; Frias and Nash, 2019), harmonize methods to sample and extract MPs from diverse matrices and subjects (Lusher et al. 2020), conceptualize the pathways in which primary and secondary MPs are leaked into the environment (Peano et al. 2020; Saling et al. 2020), and propose guidelines to ensure good practice in data collection and reporting (Cowger et al. 2021; Askham et al. 2023). Furthermore, scientists have investigated how MPs interact with the environment and those that live within it (Woods et al. 2019; Boyer et al. 2022), including humans (Ravanbakhsh et al. 2022). These efforts are critical in providing scientists with a holistic understanding of the MP problem and provide well-informed guidance to decision and policy makers. However, despite the acknowledgement of the pollutant and numerous interdisciplinary and international efforts focused on understanding MPs, their impact on the environment and human health remains unquantified. It is important to note that the impact, in the context of MP emissions, is most likely negative in that experts expect the presence of MPs in the environment to be harmful for the environment or human health. Despite the lack of quantified damage (Rai et al. 2021; Gontard et al. 2022) there is a consensus in the literature that MP's have the potential to be harmful to human health (Corella-Puertas et al. 2022). Intake pathways of MP's can be from drinking water, ingestion of plastic containing seafood or inhalation of microplastics in the atmosphere (Kawecki & Nowack 2019; Corella-Puertas et al. 2022) and have also been found present as dust in dwellings (Kawecki & Nowack 2022). MP's travel from the Technosphere to the biosphere through many pathways (Kawecki & Nowack 2019). Additionally, the MPs dissipate through several natural compartments (land, water, air, ocean) and the exposure is often several magnitudes higher in measured concentration in each compartment (Lavoie et al. 2021). Further, there is evidence that plastic degrades from larger structures to smaller, transforming from macro plastic to micro and eventually nano plastic.

Conclusively, all MP researchers seem to agree that MPs are a complex and challenging environmental problem. While the area of study has made significant and rapid advancements, there is much work to be done before their possible risks can be fully understood (Lusher et al. 2021; Croxatto Vega et al. 2021). Fortunately, this complexity is being addressed as more areas of study involve themselves in MP research. The ongoing work to date has allowed the research community to make progress in moving from a conceptual overview of MP impacts to demonstrating the realistic consequences one should expect to see or experience if society continues with business as usual.

So far, it is evident that the presence of MPs in the environment is considered a threat to ecosystems and human health (Ravanbakhsh et al. 2022). There has been significant focus on the marine environment (Lavoie et al. 2021), however, there are increasing amounts of MP impact studies within the freshwater (Wang et al. 2022), terrestrial (Junaid et al. 2023), and atmospheric compartments (Akanyange et al. 2021). Concerns with macro- and microplastic pollution has influenced policy, as is evident by the growing number of countries which have either banned or imposed taxes on single-use plastics (Croxatto Vega et al. 2021) as well as restrictions put in place regarding the manufacturing and selling of cosmetics that contain MPs (OECD 2021) or the banning of control-release fertilizers manufactured using non-biodegradable polymers after 16 July 2026 (FAO 2021).

2.2 Polyhydroxyalkanoates

Polyhydroxyalkanoate (PHA) based biofilms biodegrade under both anerobic and aerobic conditions (Mas-Castellà et al. 1995; Altaee et al. 2016; Ong et al. 2017; Dilkes-Hoffman et al. 2019). Biodegradation is the breakdown of materials by bacteria, fungi or other biological means (Ong et al. 2017). During biodegradation the long complex polymer structures are degraded to shorter non-toxic compartments that bacteria can use as an energy feedstock (Arcos-Hernandez et al. 2012; Altaee et al. 2016). During aerobic degradation the PHA's produce CO_2 and H_2O whilst under anaerobic conditions, methane (CH_4) will also be produced. There are many factors that influence the degradation of PHA biofilms such as microbial activity, pH, nutrient content, polymer composition, surface area, polymer crystallinity, etc. (Altaee et al 2016). A common PHA is Polyhydroxybutyrate (PHB) (Altaee et al. 2016). PHB has a high crystallinity which makes the material hard to work with, rigid and brittle. The introduction of longer monomers such as hydroxyvalerate (HV) can be co-polymerised with PHB creating PHBV. When the longer HV monomer groups are co-polymerized with the PHB, the hardness, brittleness and melting temperature decreases. With a broad structural variability, PHA can be tailored with specific properties for a broad range of products with varying degradation rates depending on their intended purpose (Prambauer et al. 2019).

With its natural degradation and biomaterial origin, PHA is commonly seen as an alternative material to fossil plastic polymers for mitigating plastic leakage (Ong et al. 2017; Dilkes-Hoffman et al. 2019). However, the degradation is not instant. Dilkes-Hoffman et al. (2019) report a biodegradability rate of 1.5 to 3.5 years for a water bottle made of PHA to completely degrade. However, the properties of PHA materials in the environment highly affect its degradability, ranging from 0.1 to 0.2 years for bags and from 2.3 to 5.4 years for cutlery. As such, PHA based materials pose a risk of short-term (0.1 to 5.4 years) macro-material leakage to nature. In more general terms Dilkes-Hoffman et al. (2019) found that the degradation of PHA based materials is 0.04-0.09 $\frac{\mathrm{mg}}{\mathrm{day}*\mathrm{cm}^2}$ The degradation of PHA is slowest in marine environments, followed by soil, compost, and fastest under anaerobic conditions. As such, it is observed as a positive effect of degrading PHA's through organic waste management, such as composting or anaerobic degradation (i.e., biogas production) (Dilkes-Hoffman et al. 2019). However, it is uncertain whether PHA based materials degrade fast enough to be applied for industrialized biogas production. A shortcoming in the current literature is, however, to establish if macro material leakage of PHA can pose a danger to terrestrial, aquatic and marine species. Whilst Altaee et al. (2016) report that PHA can serve as feedstock for bacteria, fungi and other organic processes, it is unknown whether it may pose risk of entanglement as macro material residing in the environment. For micro material consequences there seems to be a general consensus in the literature that PHA materials will degrade. However, degradation of PHA does not come without any changes to the environment. Brown et al. (2023) found that the use of PHBV's in agricultural soil causes a change in soil chemistry, affecting nutrient availability. Additionally, Mo et al. (2023) report that earthworms ingest weathered mulch made of PHA. Among the observed changes were the inactivation of nitrogen and phosphorus which resulted in a decreased crop yield for corn (Brown et al. 2023). As such, the presence and degradation of PHA's in nature is not fully without negative consequences.

2.3 A Brief Overview of Life Cycle Assessment of Microplastics

LCA is a method used to analyse environmental aspects and impacts of product systems (Klöpffer & Grahl 2014). It is composed of four primary phases, namely: Goal and Scope definition, Inventory Analysis (LCI), impact assessment (LCIA), and interpretation (Klöpffer & Grahl 2014). LCA is intended for direct application in the areas of product development and improvement, strategic planning, public policy making, marketing, and more (Klöpffer & Grahl 2014). The method can be used to assess new and emerging contaminants, pollutants, and emissions, however, to do so, the phases in which LCA is composed should first be developed. Although MPs have been observed since the 1970s, they are a relatively new environmental pollutant, thus, their inclusion in LCA is currently under development.

There has been significant progress in the LCI of MPs, largely thanks to the work done in the Plastic Leak Project (PLP) by Peano et al. (2020). This has meant that the LCI part of LCA has been able to make meaningful progress.

The LCIA part of LCA is when the LCI data is classified into which impact category (or categories) the emission or resource use contributes to. The next step in LCIA is characterization, where the amount a given emission contributes to a given impact is quantified. The potential environmental impacts are calculated by the mass of the emission multiplied by the characterization factor (see Equation 1). These can be impacts like global warming potential, or acidification potential, which are examples of what are usually called mid-point impact indicators by LCA experts. The LCIA process can also include calculation of damages to Areas of Protection, like human health and ecosystem quality (often called end-point damage categories, or AoPs).

Equation 1: Calculation of environmental impacts:

$$M_j * CF_{j,i} = I_{i,j}$$

The mass (M) of pollutant (j) to nature is multiplied by the CF (characterization factor) for pollutant (j) for impact category (i) and gives the total impact (I) of pollutant (j) within impact category (i) associated with the mass of the pollutant.

LCIA relies on the development of characterization factors (CFs) in order to calculate impact. CFs are developed based on fate factors (FF), effect factors (EF), and exposure factors (XF) (see Equation 2 and Corella-Puertas et al. (2022) for a detailed explanation of CF generation).

Equation 2: Calculation of Characterization factors:

$CF = FF \times XF \times EF$

LCIA for plastic products and their alternatives cannot be carried out thoroughly without first defining XFs, EFs, FFs for both macroplastics and MPs. In recent years, researchers have developed some FFs and EFs, i.e. Saling et al. (2020) and Lavoie et al. (2021). This has then allowed for the development of CFs, i.e. for LDPE in Croxatto Vega et al. (2021) and EPS and TRWP in Corella-Puertas et al. (2022). The work on developing factors for more plastics and the different types of effects they can have on organisms is ongoing. A good overview of the relevant effects and pathways is described in the framework by Woods et al. (2021). A challenge, however, is that the identified characterization factors are in their very early stages (i.e. consider only physical impacts on biota) and cannot be used to assert ecosystem damage or negative human health effects at the current stage (Rai et al. 2021; Gontard et al. 2022). Therefore, in this study, value chain MFA is employed to map and quantify the flow of plastic pollutants as MP and macroplastic from the technosphere (a products value chain) to nature, and between environmental compartments within nature. The selected product materials are presented in the following.

2.4 Case Studies

The product groups selected were based on a product segment of interest to the project commissioner. These products are of interest for introducing their PHA based materials, as they seem to have a high rate of or high total amount of plastic leakage to nature. The selected product groups are geotextiles, control-release fertilizer film, mulch film and dolly ropes. Each of these product groups are traditionally made from fossil-based polymers. The fossil polymers used to manufacture the said product groups vary. Different fossil polymers have different impact potential (Kawecki & Nowack 2019). Similarly, macroplastics and MPs also have different impact potentials and pathways in nature (Peano et al. 2020). Therefore, it is important to assess the individual product group's MP and macroplastic leakage potentials in relation to the common polymer types when one assesses the potential impacts associated with plastic leakage (Peano et al. 2020).

In this study we quantify the plastic leakage potential from each of the mentioned product groups. The plastic leakage is then fractioned into likely MP and macroplastic quantities. Depending on the product type the leakage pathway to nature is determined for each product group's value chain stage. The estimated plastic leakage at MP and macro level to a given environmental compartment can then be multiplied with the relevant CF for the polymer type used for the specified product group. Each included product group is significantly different in material composition, intended use and thus it's potential plastic leakage to nature. A short description of each of the product groups to be replaceable by the use of PHA's and PHBV's follows.

2.4.1 Mulch Films

Mulch film is a product group that is used on agricultural soil intended to create an artificial atmosphere and protection against pests and drying. The most common fossil polymer used for mulch film purposes is low density polyethylene (LDPE) (Bauchmüller et al. 2021; Croxatto Vega et al. 2021; FAO 2021). It is estimated that more than 2 million tonnes of mulch film are used each year in agriculture, globally (FAO 2021). For the EU the tonnage in 2020 was about 75 000 tons with a leakage potential of roughly 10% (Bauchmüller et al. 2021). Croxatto Vega et al. (2021) show that the weather conditions affect the fate of plastic much-film on agricultural fields. Their study provides factors for plastic particles entering the air in Denmark from 6% to 23% depending on wind conditions (high wind means the higher percentage entering the air). Correspondingly, high wind means that only 77% of particles deposit on the ground, compared to 94% in low wind conditions. In Switzerland, another study estimated that 100% of MPs from LDPE used in the agricultural sector ended up in the soil (Kawecki & Nowack 2019). However, in line with Peano et al. (2020) some releases of MP from land to water compartments have been included to account for run-off of microplastic from land via freshwater. Geography, sampling methods, estimation practices among other metadata are important factors when sampling and evaluating the risk of plastics ending up in the environment (Askham et al. 2023).

2.4.2 Geotextiles

Geotextiles are products made of permeable fabrics used primarily for ground stabilization, drainage and reinforcing purposes for construction of buildings, roads and land or mudslide prevention. Therefore, geotextiles are a product group with both short-term and long-term applications (Prambauer et al. 2020; Hasan et al. 2020). Short-term applications of geotextiles are soil erosion prevention. For short-term geotextile applications, onsite degradation within a few years would reduce the need for recollection and mitigate MP and macroplastic leakage to the surrounding land and waterbodies. Therefore, natural fibers or biodegradable materials can be a good source material for short-term geotextile applications. The benefit of using biodegradable materials such as PHBV's is that these perform well regarding water diffusion. For natural fibers, the water absorption can render the product several hundred percent heavier than its dry weight which is a risk element that must be considered in soil erosion areas (Prambauer et al. 2019). The drawback of using PHA's for geotextile purposes is its apparent cost and manufacturable quantity, based on 2019 figures, which make its use as a prominent geotextile unfeasible. However, future increases in demand of PHA's may change the available material quantities and future prices (Prambauer et al. 2019). Degradable short-lived materials are not suitable for long term applications such as in road and building constructions because the longevity of the material is key to its function. As the PHA biodegradable materials are relatively short lived it is not reasonable to assume that PHA's can theoretically replace fossil-based raw materials for all geotextiles. The theoretical replacement potential for using degradable materials for geotextile purposes is estimated to be about 50% of existing geotextiles. In 2019 2% of geotextiles were already fiber based (Prambauer et al. 2019). As fiber based and biodegradable geotextile materials share functionality and usages it is likely that the current 2% of geotextiles must be subtracted from the estimated biodegradable material replacement potential of geotextiles. In 2021 the amount of geotextiles in use in the EU was about 100 000 tonnes (Bachmüller et al. 2021).

2.4.3 Control-Release Fertilizer

Control-release fertilizer is a polymer film product that ensures that a fertilizer is released over time, ensuring long term fertilization of agricultural fields, and can be used to mitigate nutrient runoff. The most commonly used fossil polymer for this application is polyurethane (PU) which is intentionally left in the soil (Bauchmüller et al. 2021). It was estimated from 2018 values, that polymer coated fertilizers contributed approximately 0.1 million tonnes to the plastic used in agriculture globally (FAO 2021). With plowing and runoff from agricultural fields the MPs from the control-release fertilizer film can be transported to freshwater systems and into the ocean, however, the necessary specific data on run-off of this type of plastic particles from land to the ocean is lacking in the current literature. Additionally, whilst agricultural retention of LDPE, HDPE, PET, PP (Kawecki & Nowack et al. 2019) and tire wear (Siegfried et al. 2017) have been observed, no such data were available for PU. As PU has a similar density (1.28 to 1.66 g / cm^3) (MatWeb, 2023) to PET (1.4 g/cm^3) (ScienceDirect, 2023) this study uses data suitable for PET microplastics as a proxy for PU used as film in control-release fertilizers. The annual consumption of control-release fertilizer plastics was about 10 000 tonnes within the EU (Bauchmüller et al. 2021). Of the total tonnage of control-release fertilizer film, 100% is left in the fields and lost to nature. Based on the small size of control-release fertilizer it is assumed that 100% of the plastic leakage is as MPs.

2.4.4 Dolly Ropes

Dolly ropes are a product that is used to protect bottom trawling nets against wear and tear against the seabed. Dolly ropes are subjected to this wear and tear causing plastic leakage to occur at the seafloor in the marine environment. Dolly ropes are primarily made from polypropylene (PP) (99%) (Bauchmüller et al. 2021). Several sources refer to the use and wear and tear of dolly ropes in the sea. It is estimated that a trawler uses between 325 kg and 3 500 kg of dolly rope per year (FAO 2021). Veiga et al. (2016) discloses a plastic leakage to sea during use at 10-25% whereas the Bauchmüller et al. (2021) reports 50% leakage. This deviation in estimates for loss reflects that there are significant uncertainties associated with the wear and tear of dolly ropes. As the wear and tear occurs during the use phase in the oceanic compartment, there is less uncertainty regarding the natural compartment pathways of dolly ropes. The density of the material worn down to MP and macroplastic is a key parameter that determines the fate and retention time of a polymer in a water body (Lavoie et al. 2021; Corella-Puertas et al. 2022). For PP, the density is 0.9 g/cm^3 (Greene 2021) which is less than for water (1.03 g/cm^3) (Corella-Puertas et al. 2022) and is likely to remain in the ocean instead of depositing in the sediment.

The annual tonnage of dolly ropes is about 1 000 tonnes in the EU for the year 2021 (Bauchmüller et al. 2021).

3 Keyword Co-Occurrence Analysis

3.1 Bibliometric Network Analysis

Bibliometric network analysis is a tool used to efficiently extract relevant data from a scientific literature search. This form of analysis allows for a comprehensive overview of several bibliographic data points including authors, journals, and keywords (van Eck and Waltman, 2022). The keyword co-occurrence analysis can be a useful tool for establishing relevant topics and study areas and their direct and indirect relationship with other topics and study areas. This is because co-occurrence means that two terms appear alongside one another in a given scientific publication. Investigating these relationships can provide an overview of interdisciplinary collaboration and key areas of interest in ongoing research, as well as highlight possible knowledge gaps or areas for improvement. The keyword co-occurrence analysis results in a network map in which keywords are shown as bubbles and links (which indicate a co-occurrence) are shown as lines between the bubbles.

3.2 Method

A literature search was carried out using Scopus on 22/11/2022 using search term: "microplastic* AND impact* AND assessment". All citation information, bibliographical information and abstract & keywords results were exported as .csv to be uploaded into VosViewer version 1.6.18 bibliometric network analysis software. Using VosViewer, the keyword co-occurrence analysis was selected, and all keywords were exported into excel to create a thesaurus. The thesaurus categorized similar terms or terms which were spelled in different ways or abbreviated. For example, a generic keyword like "environmental compartment" or "environmental factors" would be categorized under "environment". In addition, general terms, such as "article", were removed from the keyword list to allow for better visibility in the network map (See Appendix 1 for thesaurus). Impact was the focus of the literature review, therefore, all keywords containing "impact" were searched for in the final network map. While "impact" and "human impact" were included in the final network map, they had no direct links to any other keywords (i.e. no lines connecting their bubbles to other bubbles). However, the keyword "environmental impact" did contain visible links to other keywords in the network map, therefore, it was selected for a detailed keyword analysis which involved selecting the connecting keywords to reveal their respective links. Finally, while the terms "plastic" and "microplastic" are shown in the network map, they were excluded from the keyword analyses because they inherently have the strongest link to other keywords due to their presence in the Scopus search term.

3.3 Results

The keyword co-occurrence analysis initially returned 8,784 keywords, which was reduced to 4,107 after the thesaurus file was applied (see Appendix 1 for all keywords). The network map displays the top 1,000 keywords (with respect to link value) to allow for better visibility (Figure 1). It is important to note that the colors in the network map are assigned by the VosViewer software based on its sorting algorithm, therefore, they are arbitrary for the purposes of our investigation which focuses on the size of the bubbles and the presence of lines connecting keywords. The keyword "environmental impact" (Figure 2) demonstrated links to 56 other keywords, listed in Table 1.

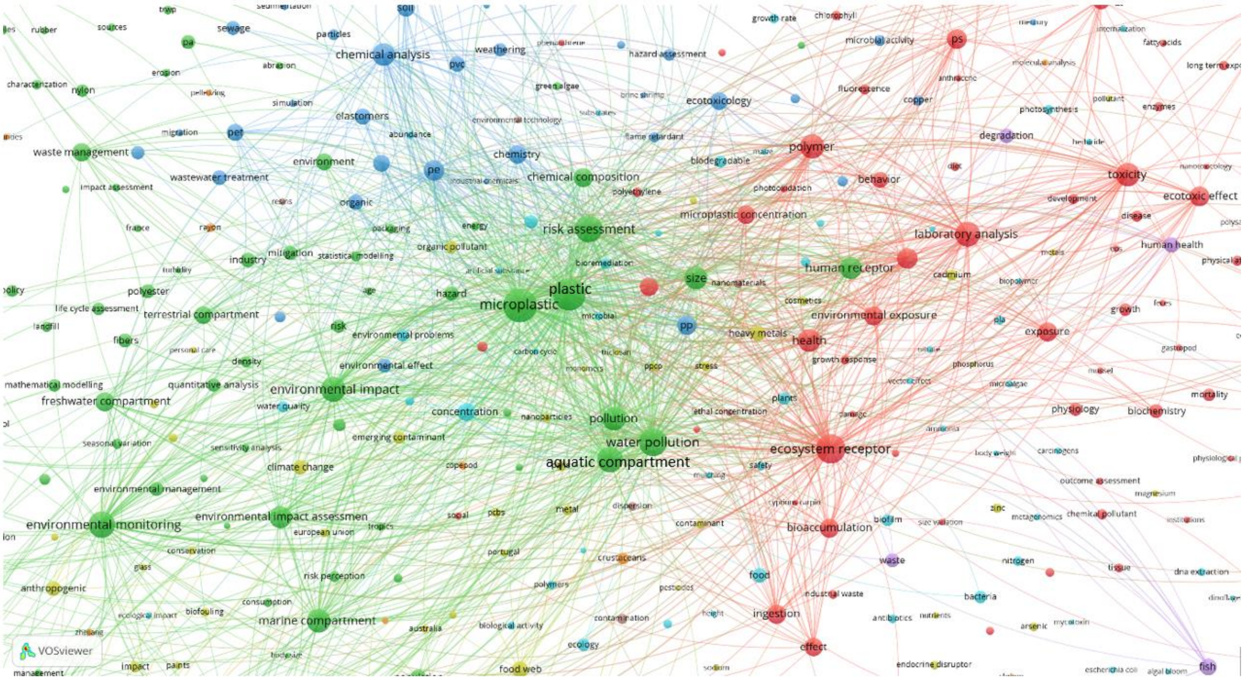


Figure 1. Keyword Co-Occurrence Network Map showing all keywords after thesaurus file was applied (zoomed in).

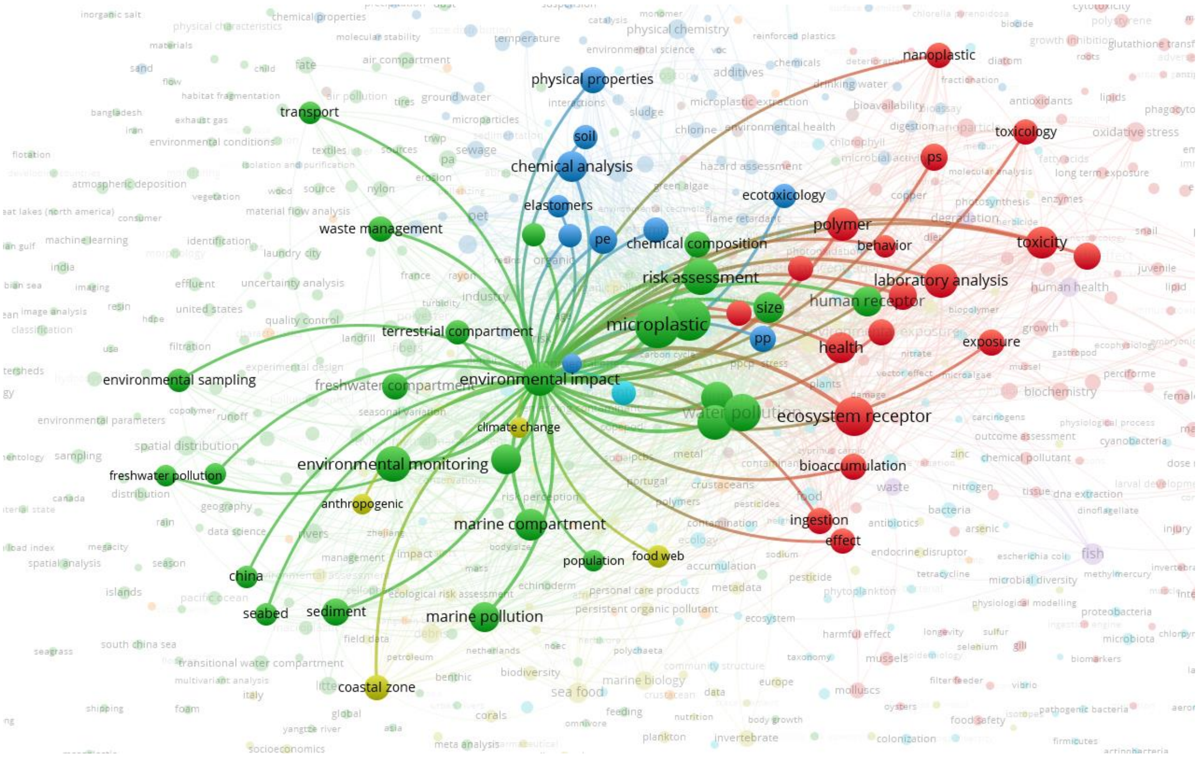


Figure 2. *Keyword Co-Occurrence Network Map showing keywords connected to "environmental impact".*

Table 1: Keywords with VosViewer links to "environmental impact" and respective link strengths. Link strength indicates the prevalence of the co-occurrence, i.e. larger link strength = more prevalent co-occurrence.

Keyword	Link Strength to "environmental impact"
Ecosystem Receptor	214
Water Pollution	169
Risk Assessment	160
Environmental Monitoring	142
Aquatic Compartment	134
Pollution	110
Marine Compartment	110
Laboratory Analysis	99
Marine Pollution	96
Polymer	94
Toxicity	93
Size	86
Chemical Analysis	82
Human Receptor	80
Health	80
Drugs	72
Environmental Impact Assessment	61
Sediment	60
Bioaccumulation	59
Freshwater Compartment	58
Ecotoxicology	57
Waste Management	57
PE	57
Chemical Composition	57
Terrestrial Compartment	52
Microplastic Concentration	51
Soil	50
Environmental Exposure	49
Ingestion	49
Coastal Zone	48
Physical Properties	46

PS	45
PP	45
Environment	44
Environmental Risk Assessment	44
Microplastic Composition	44
Chemistry	43
Effect	42
Concentration	41
Behavior	40
Toxicology	40
Exposure	40
Food Web	40
Ecotoxic Effect	39
Environmental Sampling	38
Elastomers	38
Transport	37
Seabed	37
Climate Change	34
Environmental Effect	34
Population	33
Life Cycle Inventory	33
China	33
Nanoplastic	31
Freshwater Pollution	31
Anthropogenic	30

3.4 Discussion

The results of the keyword analysis demonstrate the numerous topics that are discussed in the literature in the context of environmental impact. For example, Table 1 shows that the term “ecosystem receptor” has the largest link strength to “environmental impact”. This indicates that there is a relatively large discussion in the scientific literature regarding the presence of MPs in the ecosystem and how this contributes to a possible environmental damage potential. The thesaurus term “ecosystem receptor” includes numerous related terms, from microalgae to wild animals (see Appendix 1 for all included terms), indicating that studies on multiple facets of the environment are being investigated, acknowledging that the presence of MPs and their interaction with the environment are likely to contribute to environmental impacts worth exploring. Furthermore, the importance of “toxicity” with respect to “environmental impact” indicates that there is evidence to support that researchers aim to demonstrate the toxic effects of MPs on the environment.

A detailed keyword analysis was carried out to better understand which synergies exist between each of the 56 terms co-occurring with “environmental impact”. This analysis highlighted critical data gaps relevant to the development of LCI of MPs, and therefore LCIA of MPs. For example, while experts have discussed and defined the pathways for MP emissions from the terrestrial compartment (including soil) to the marine compartment (Corella-Puertas et al. 2022), the literature shows that there is relatively little focus on “soil” and “terrestrial compartment” with respect to topics such as “environmental monitoring”, “environmental impact assessment”, “toxicity”, and “toxicology”. In addition, “terrestrial compartment” and “marine compartment” did not show any co-occurrence. The full analysis can be found in Appendix 2, highlighting numerous instances in which there is no co-occurrence among areas of study that share important relationships for understanding MP impacts.

4 Literature Review

Literature review is an effective form of qualitative analysis to demonstrate the status of an area of study. In the present study, literature review was critical in demonstrating the current possibilities of impact assessment of MPs and highlighting existing data gaps. As described above, impact assessment of MPs (particularly LCIA of MPs) is an area of study under methodological development. However, given the continuous effort to advance the field over recent years, significant progress has been made allowing for experts to observe and define probable impacts from MPs. Therefore, the following outlines new and ongoing work in LCIA of MPs to help guide future efforts and to validate current hypotheses.

4.1 Method

The Scopus search described in section 3.1 was also used in the literature review. A search for "impact assessment" was carried out in the exported bibliographic information. The relevant articles from the "impact assessment" search were read thoroughly to extract information related to LCI, EFs, FFs, CFs, LCIA, and LCA. These categories were selected with reference to Askham et al. (2023), based on the MarILCA framework (Woods et al. 2021). In addition, it was noted which environmental compartment was the focus, i.e. air, terrestrial, freshwater, or ocean/sea. These data were also used to help populate the MFA described in Chapter 5, where supplementary information was required.

4.2 Results

The Scopus search on 22nd November 2022 resulted in 838 documents. Of the 838 documents, 7 mentioned LCIA. Of these 7 articles, 6 were focused on LCIA of Microplastics, namely Loubet et al. 2022, Lavoie et al. 2022, Pauna and Askham 2022, Corella-Puertas et al. 2022, Croxatto Vega et al. 2021, and Saling et al. 2020. Table 2 shows the relevant information extracted from the 6 selected articles.

Table 2. Summary of relevant content provided by articles discussing LCIA.

Source	LCI	Fate Factor(s)	Effect Factor(s)	Character-ization Factor(s)	LCIA	LCA	Air, Land, Fresh-water, Marine	MP Related Impacts Described
Saling et al. 2020	No	Yes (drived from disposal statistics)	Yes (plastic pellet, fragment, and fiber, EC50, LC50)	Yes (LDPE and PP)	No	No	Marine	Physical impacts (entanglement and ingestion), chemical impacts (sorption of chemicals from the environment), and plastic debris impacts (transport of alien species and pathogens)
Croxatto Vega et al. 2021	Yes	No	No	Yes (in kg NMVOC eq/kg, kg CO2eq/kg, PM2.5 eq/kg, and DALYs)	Yes	Yes	Air/Land	Physical impacts on human health (inhalation, accumulation (in lung tissue), vectoring of heavy metals). Physical impacts on marine organisms (cell metabolism, gene expression, cell signaling, digestive, neurological, immune, growth, reproduction, feeding). Toxic effects on organisms (cytotoxicity, estrogenicity, antiandrogenicity, oxidative stress)
Lavoie et al. 2021	No	No	Yes (physical effects on biota (EC50, HC50))	No	No	No	Marine	Physical effects on biota
Corella-Puertas et al. 2022	No (refer to plastic leak project (Peano et al. 2020))	Yes (degradation rate/sedimentation rate)	Yes (physical effects on biota from Lavoie et al. 2021)	Yes (EPS, TRWP)	No	No	Marine	Physical effects on biota
Loubet et al. 2022	Yes	No, but Pre-Fate (Final Release Rate)	No	No	No	No	Marine	Socioeconomic and environmental impacts (contamination of fish with ingested plastics, restricted catch due to litter in nets, vessel damage and staff downtime, reduced earnings and lost fishing time), physical impacts on ecosystem quality, physical impacts on marine wildlife (entanglement, ensnare, ingestion)
Pauna and Askham 2022	No (some data related to path-ways)	No	No	No	No	No	Marine	n/a

4.3 Discussion

While 6 articles from the literature review had a focus on, and even carried out LCIA of MPs, the main contributions were the discussions related to potential damages of MPs, particularly those with respect to polymer type. This is likely because there are yet to exist standardized CFs for LCIA of MPs. Researchers, including Corella-Puertas et al. (2022) are currently working to develop CFs for the marine compartment, however, this is not the final sink, or location of final retention, for all MPs. Therefore, it is currently only feasible to carry out simplified LCIA for MPs, on a case-by-case basis, limiting the amount of usable data to be used for a regionalized LCIA. This is also being addressed through data guidelines presented in Askham et al. (2023), which have been developed by experts in Risk Assessment, LCA, Ecotoxicology, and MP Sampling and Extraction. These guidelines have been published to encourage documentation and provision of relevant data from industry related to petroleum-based and bio-based plastics, as well as to ensure that they are applicable for use in LCI and LCIA. The guidelines are also relevant for MP researchers to ensure that adequate metadata are collected when sampling and experimentation is carried out. Despite data needs, the literature review demonstrated that there are ways to estimate probable environmental impacts related to MP emissions. In addition, there are laboratory assessments that have demonstrated toxic effects on relevant species. Furthermore, MPs have been found in human tissue. Therefore, there is less of a question of: are MPs dangerous to the environment and human health? but rather: how dangerous are MPs to the environment and human health?

5 Material Flow Analysis

5.1 Background

Material flow analysis (MFA) is a common method used to assess the flows of materials, energy or substances to, through and out of a defined system (Brunner & Rechberger 2004). An MFA requires both qualitative data and quantitative data. The qualitative data describes how the system activities relate to each other within the defined system boundary and in relation to the outside compartments. The quantitative data describes or calculates the quantities of the interactive flows between activities within the defined system and their relation to the outside world. The overall aim of an MFA is to provide a holistic understanding of how a system works and interacts within and outside itself (Brunner & Rechberger 2004). When MFA is applied for addressing the flow of environmental pollutants and/or product value chain flows it is regarded as an environmental field of science (Elia et al. 2017). Because an MFA can be used to trace a product’s value chain flows and estimate the flow of pollutants to nature it is compatible with LCA (Elia et al. 2017; Pauna & Askham 2022).

There are two ways of integrating an LCA with an MFA. One way is to use the MFA to estimate the flow of a pollutant and then multiply the mass flow data for a given pollutant with a CF. A CF factor is a factor that determines the environmental damage of a pollutant within an impact category (i.e., global warming potential), as described in section 2.3 of this report. This manner of combining MFA with LCA for estimating the environmental damage associated to plastic leakage is an emerging field of science still in its infancy. Currently there are no complete CFs published for microplastic or MP leakage to nature.

Equation 2 (previously shown in 2.3): Equation for integrating MFA with LCA

$$M_j * CF_{j,i} = I_{i,j}$$

The mass (M) of pollutant (j) to nature is multiplied by the CF (characterization factor) for pollutant (j) for impact category (i) and gives the total impact (I) of pollutant (j) within impact category (i) associated to the mass of the pollutant. The sum mass of a pollutant to nature is the net impact emissions associated to an activity or product value chain. The net impact of emissions is a common finding from an LCA study.

A more developed way of combining MFA and LCA is to estimate the foreground system flows of a product's value chain, from cradle to grave. The foreground product flow is quantified using MFA which then can be multiplied with an LCI database. The LCI database provides emission factors for each value chain activity per unit of product mass flow going to or from it. This way of combining MFA with LCA not only lacks the characterization of plastic leakage damage (Rai et al. 2021) but there is also a lack LCI databases describing MP and macro plastic leakage potential associated with value chain activities (Gontard et al. 2022). Traditionally, an MFA would need to track the flow of a material or substance until it experiences a change in that flow, i.e., division of a flow into two flows. Areas where a material or substance flow experiences a change are called compartments. A compartment can be an anthropogenic activity within the technosphere or a natural environment within the biosphere.

The traditional manner of carrying out an MFA would be limited to addressing only product related plastic leakage potential (Brunner & Rechberger 2004). To combine MFA with traditional LCA method practice one must also take into account indirect emission sources associated to the value chain activities needed to cause a change to the product in question. There is also a need for distinguishing plastic leakage into MP (<5mm) and macroplastic (>5mm). There is strong evidence that there are vastly different environmental impacts associated with MP (<5 mm) and macroplastics (>5mm) (Minderoo Foundation, 2022). Additionally, the path and means that MPs and macroplastics transport between environmental compartments differs (Kawecki & Nowack 2019; Peano et al. 2020). Therefore, we distinguish in this study between MP and macroplastic leakage potential.

There is strong evidence that different fossil polymers pose different environmental effects (Lavoie et al. 2021). The environmental effect of a pollutant is often called the Effect Factor (EF) and is an integral part of developing a CF (see section 2.3 and Maga et al. 2023 in press; Peano et al. 2020). Therefore, this study includes identifying which fossil polymers are commonly used in the product groups we address.

5.2 Method

In MFA and LCA it is important to present a flowchart depicting the system boundary comprising included and key excluded activities and processes (Brunner & Rechberger 2004). The assessed technosphere systems (Figure 3) include seven key life cycle stages that represent common life cycle value chain activities. There can be some plastic littering associated with oil extraction activity (i.e. oil platform activities offshore), but in this study this is considered negligible, as the plastic product is not yet produced and there is a lack of data for plastic littering potential from oil extraction. The flow of oil to plastic polymer production is considered as the flow of raw material entering the system. The following upstream life cycle activities are Polymer production (A2) and Manufacturing of product (A3). The use phase (A4) represents the point at which the polymers are in active use. The following product stages are the downstream activities Waste management (A5), Landfilling of plastics (A6) and Recycling of plastics (A7). The arrows depict the direction of the fossil polymer mass flows within the system boundary and the dotted line arrows depict the plastic pollution potential from each value chain activity.

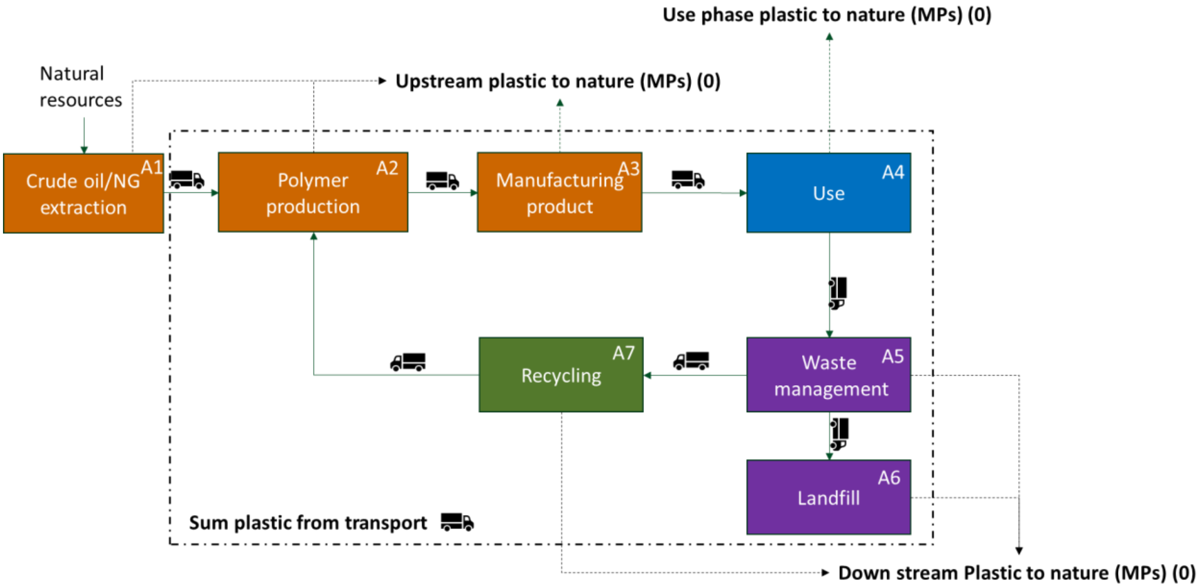


Figure 3: Process flowchart over included technosphere compartments

The MFA approach used is stock driven MFA. The point of reference is the Use phase (A4), and the stock is the market amount of a given product group. In this study the region assessed is Europe and the year is 2021. Marked masses of the included product groups are based on the Bauchmüller et al. (2021). The product related plastic leakage to nature is established by a relative share (transfer coefficient) of the product material that is plastic that can be leaked and the activity related plastic leakage. These transfer coefficients are collected from various sources. The most important sources for the estimation of transfer coefficients from technosphere to nature, and between environmental compartments are (Kawecki & Nowack 2019 and Peano et al. 2020).

Two types of plastic leakage to nature from a value chain activity are defined: product related plastic leakage and activity related plastic leakage. The product related plastic leakage to nature is plastics that are used as part of the product in question. An example of product related plastic to nature is PU film used to contain control-release fertilizer that remains in the soil after use. Activity related plastic leakage is plastics that are in use within the activity necessary to fulfill the function of that activity and are thus plastics indirectly associated to the product in question. An example of an activity related to plastic leakage is rubber tire wear from lorry freight. There is no activity related plastic leakage found for the value chain process activities A1-A7. However, there are activity related plastic leakage associated with road transport due to tire wear (Peano et al. 2020). The total plastic to nature is therefore product related plastic leakage plus transportation related tire wear emissions. The sum of transfer coefficients determining the product flows should always equate to 100% to ensure mass balance consistency (Brunner & Rechberger 2004). The activity related transfer coefficients between each value chain activity, the technosphere and nature, and each environmental compartment is presented in detail in Appendix 3.

The distinction between product and activity related plastic to nature is important for two reasons. Firstly, the product and activity-related plastic polymers are likely to be of different materials, i.e., PP from the product and rubber from the tire wear. Secondly, when comparing products, the product polymer might change whereas the activity-related emissions and type often remain unaffected. The activity related plastic leakage might therefore be a valid environmental concern for products that otherwise mitigate product related plastic leakage.

The method used to assess the environmental implications of applying PHA instead of fossil-based polymers is to estimate the difference in plastic leak potential in terms of MP and macroplastic that would otherwise be caused by the fossil-based materials of the specified product group. The activity related plastic leakage potential (i.e., tire and road wear particles (TRWP)) remains the same regardless of the use of PHA or fossil-based materials for each product group, as there is assumed no change in road transport distances or mass transported.

Toxicity assessments in LCIA commonly consider the environment in terms of box models, i.e. with environmental compartments; three such compartments are used in this study: land, freshwater, and the ocean. The plastic leakage emissions for each product group are considered for each different environmental compartment. For plastic leakage to land and freshwater there is additionally a transfer to the ocean. The reality is much more complex than the three environmental compartments defined. A challenge with quantifying the flow of MP and macroplastics through natural compartments is that these polymers are persistent pollutants. This means that MPs and macroplastics reside for a certain time period within one environmental compartment before partially or fully transporting to another (Han et al. 2022). Additionally, factors such as topography, rainfall, wind patterns, basin size and vegetation severely affect the flow and retention of plastics in the environment (Siegfried et al. 2017; Han et al. 2022). Similar challenges are evident in the transfer between the technosphere and nature where local wastewater treatment plants have different efficiencies for capturing MPs (Siegfried et al. 2017). To calculate the transfer coefficient from the technosphere to the river mouth for a given material type (i) $FE_{riv,i}$ Siegfried et al. (2017) propose an equation (Equation) that takes what is not retained in the land (1-Ret,i) multiplied by the portion of river water that is not consumed by the technosphere $(1 - FQrem,_i)$.

Equation 3: Transfer coefficient from land and river to the ocean following the method described in Siegfried et al. (2017)

$$FE_{riv,i} = (1 - Ret,_i) * (1 - FQrem,_i)$$

Siegfried et al. (2017) report TRWP retention in the sediments from two literature sources at 75% (Sundt et al. 2014) and 90% (Verschoor et al. 2016). Additionally, Siegfried et al. (2017) report that the MP removed by sewage treatment varies greatly between countries. For Germany they found an MP removal rate of about 95% whereas for the Netherlands it was 50% based on data from the year 2000 from the data base Global NEWS input data. However, none of the product groups included in this study is likely to enter the national wastewater system as they are primarily emitted directly to land (control-release fertilizer, mulch film and geotextiles) or the ocean (dolly ropes).

In this study we have chosen a crude nature system distinguishing between land, rivers and lakes, and the ocean as separate environmental compartments, Figure 4. A crude partitioning of nature pathways is currently common practice in the wider literature (Peano et al. 2020), but further research will eventually lead to more detailed environmental compartment systems taking into account terrain, socio-technological conditions such as wastewater treatment technology and localized natural barriers and basin sizes. Another challenge with plastic in nature is that MP and macroplastics enter and move at different rates through nature based on the polymer type and product category (as well as additives used). Currently generalized transfer ratios are suggested for product MP, macroplastics and TRWP between the environmental compartments.

Table 3: Product related environmental transfer coefficients

Transfer route	Transfer coefficient (%)	Litterature source
MP Land → Water/(Ocean)	40 %[1]	Peano et al. (2020)
MP Land → stored in soil	60%	Peano et al. (2020)
Macro land → Water/(Ocean)	25 %	Peano et al. (2020)
Macro land → stored in soil	75%	Peano et al. (2020)
MP water → Ocean	70 %	Corella-Puertas et al. (2022)
MP water → sediments	30%	Corella-Puertas et al. (2022)
Macro water → Ocean	100 %	Peano et al. (2020)

[1] Peano et al. (2020) does not distinguish between water and ocean, however they state that 100% MPs within Water/ocean remains there. Therefore, we assume that water is the pathway MPs take on from land, particularly inland areas, to transport to the ocean. Of the transport from water to ocean 30% is reported stored in the sediments. The same mode of transport between environmental compartments are assumed for microplastics, although with different transfer coefficients.

For tire wear MPs about 69,9%, 17,1 and 14% were estimated to go directly to the respective environmental compartments land, freshwater and incineration. There were no data found regarding the rate of MP transferal from land to freshwater, so the 40% found in Peano et al. (2020) (Table 3) was used to estimate this flow. For TRWP, 75 to 90% were found to be retained within freshwater sediments (Siegfried et al. 2017).

A flow not considered in this study is MPs leaking from macroplastic due to wear and tear and photodegradation (Siegfried et al. 2017; Peano et al. 2020). However, it is challenging to find good data on the degradation of macroplastic to MP. The macroplastic may have undergone several rounds of exposure to various species within the biosphere before it becomes MP, in which it can be a reoccurring hazard affecting a cascade of species. These processes can take place over a long timescale.

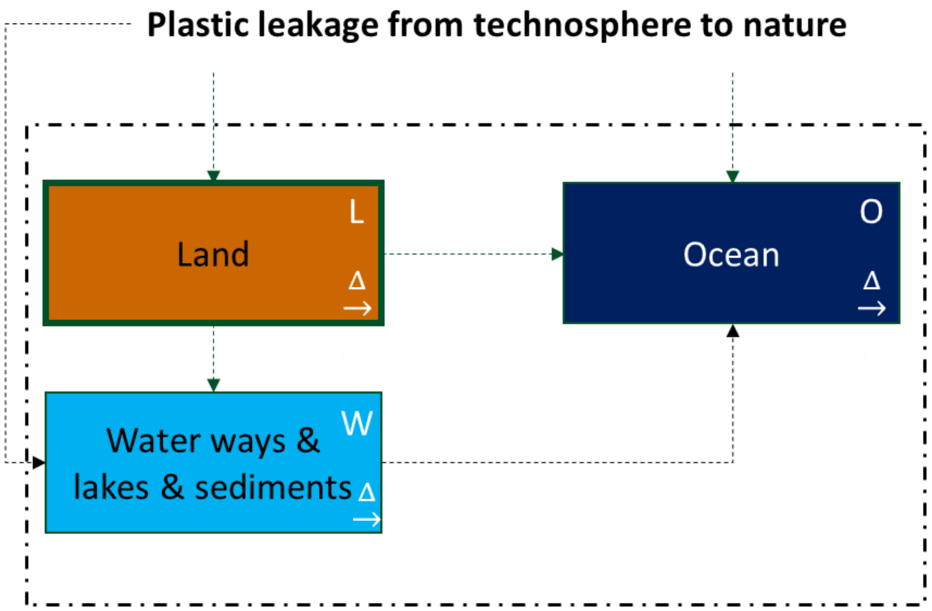


Figure 4: Process activity flow chart of plastics in nature

5.3 Results

Tracing the flow of materials through the foreground life cycle system is a key component for establishing an impact assessment. The MFA results demonstrate that the greatest potential for plastics leakage in general, MP and macroplastics combined, for each of the investigated product categories is geotextiles (Figure 5) when considering the annual use of each material in Europe, year 2021. This is followed by mulch film, control-release fertilizer, and lastly dolly ropes. For all fossil product groups, the use phase is the main plastic polluting source. However, for mulch film which has a high collection to waste rate, waste mismanagement is also a major source for plastic leakage. PHA's barely demonstrate any plastic leakage potential compared to the fossil materials for all product groups combined. This is largely linked to the assumption that the degradation time from macro to microplastic is less than 6 years and that the degraded PHA is considered consumable for microorganisms, fungi and bacteria (see section 2.2). To determine the impacts of plastic leakage one needs to distinguish between MP and macroplastic and determine which environmental compartment they reside in.

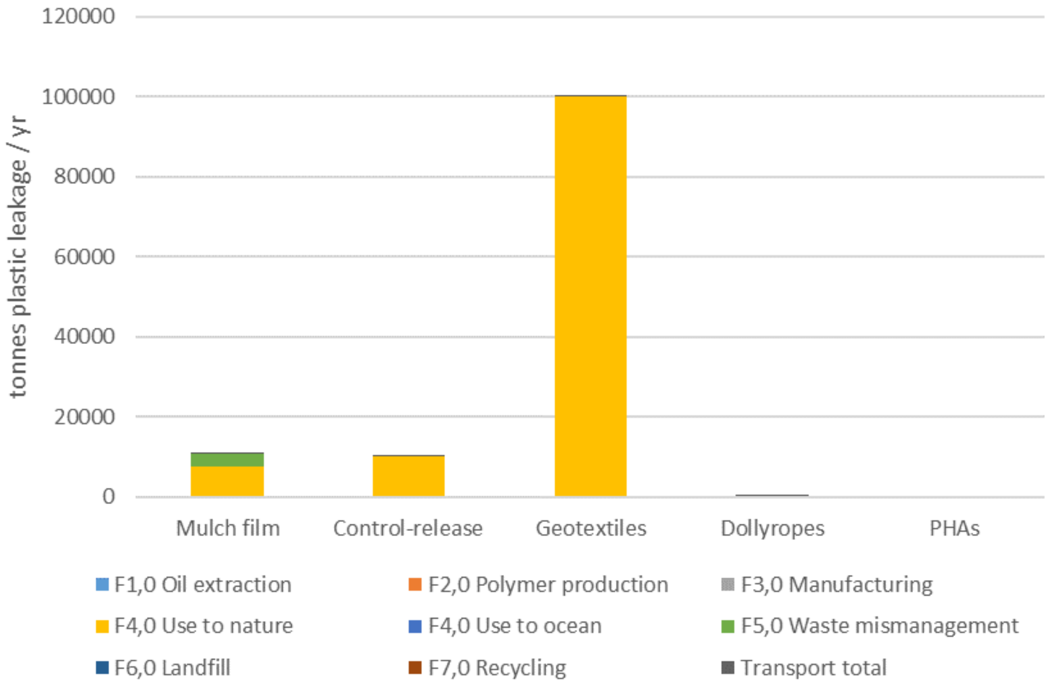


Figure 5: Tonnes of plastic leakage to nature potential in EU28 2021

The results for MP leakage potential shows that the breakdown into MP plastic pollution potential can change the relevant importance of a product category vastly compared to the initial release (Figure 6). This is in part because different materials have different ratios of plastics becoming MP and macroplastic (Appendix 3). Additionally, the transfer of persistent pollutants within a long timeframe means that the same pollutant occurs in several environmental compartments. For MPs to water, a portion is also quantified on land. For MPs to the ocean, some are both quantified on land and in the water, following the transfer coefficient of Table 3. The results show that control-release fertilizer is the product group with the highest MP potential for each environmental compartment assessed. Next come geotextiles, mulch-film, and dollyropes. PHAs demonstrate barely traceable amounts of MPs, amounting to about 7 tonnes in total if it were to replace all the fossil materials in all of the product groups. For macroplastics, the plastic leakage to each environmental compartment is vastly different.

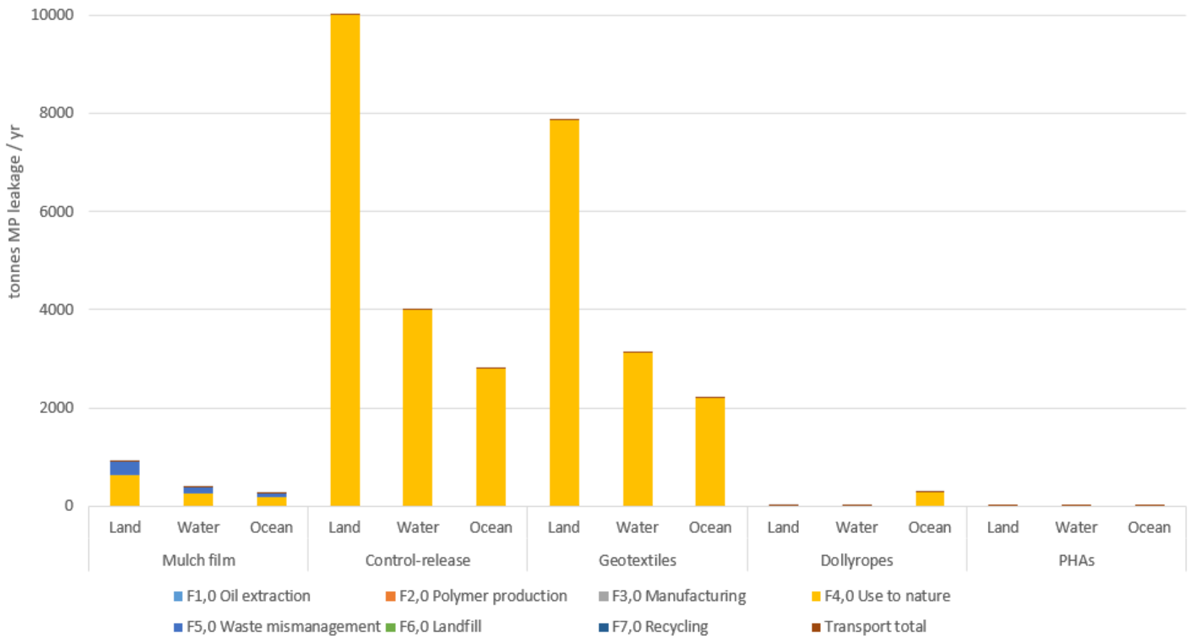


Figure 6: MP leakage potential associated with annual consumption of product groups in EU, 2021

For the products studied, macroplastic leakage goes predominantly directly to the land compartment. However, it is only geotextiles and mulch film that can lead to any significant quantity of macroplastic pollution (Figure 7). Geotextiles pose the largest risk of macroplastic pollution. However, most geotextiles are dug into the ground and remain under buildings, roads and soils, which makes them less likely to be mobilized and more likely to stay at or near their location of use. As such it is assumed that geotextile macroplastics do not leak to the other environmental compartments. Mulch film is used on the soil surface, and it is assumed that macroplastics from mulch film follow the general transport route for macroplastics found in the literature. This means that mulch film has a larger potential to cause macroplastic leakage to water and the ocean than geotextiles despite its much lower market volume and initial release to land. The alternative PHA's do not pose any risk of macroplastic contamination after 6 years, as such the emissions are excluded from this study. Whilst these findings demonstrate the accumulated potential presence of MP and macroplastic per compartment, they do not show the mitigation potential of using PHA's. The mitigation potential of using PHA based materials is presented in the following.

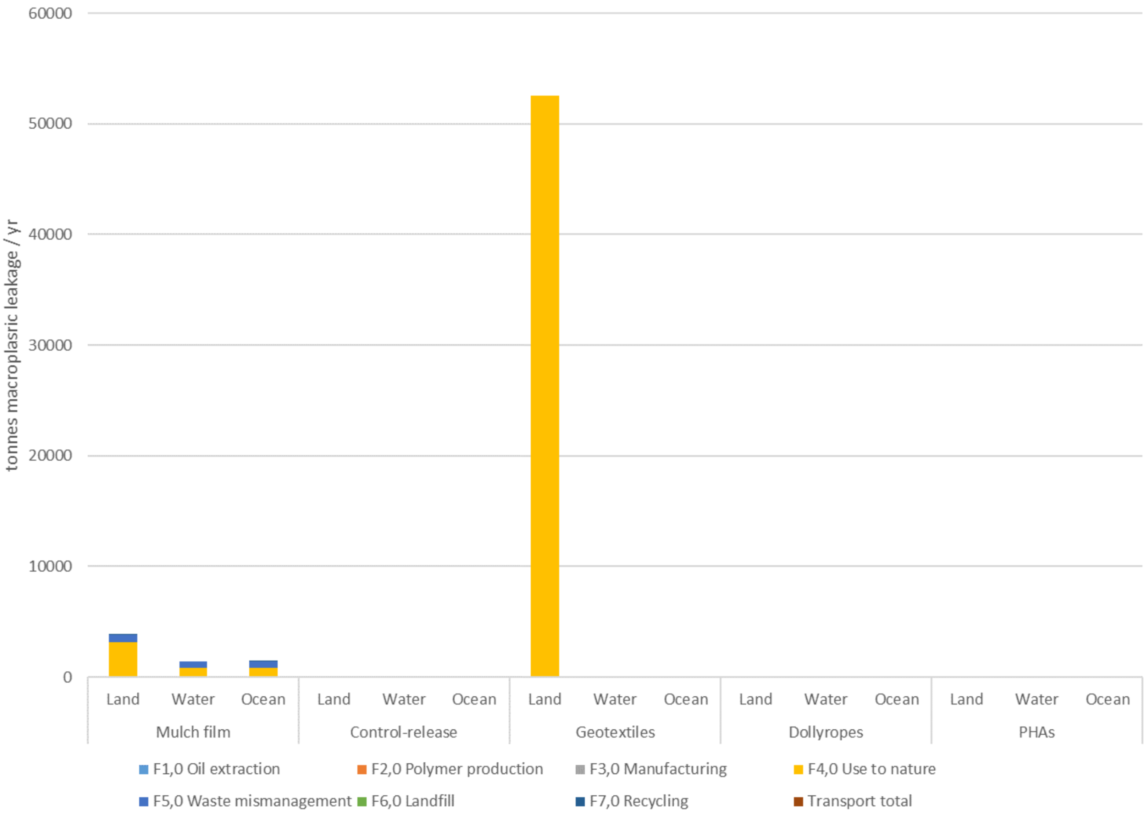
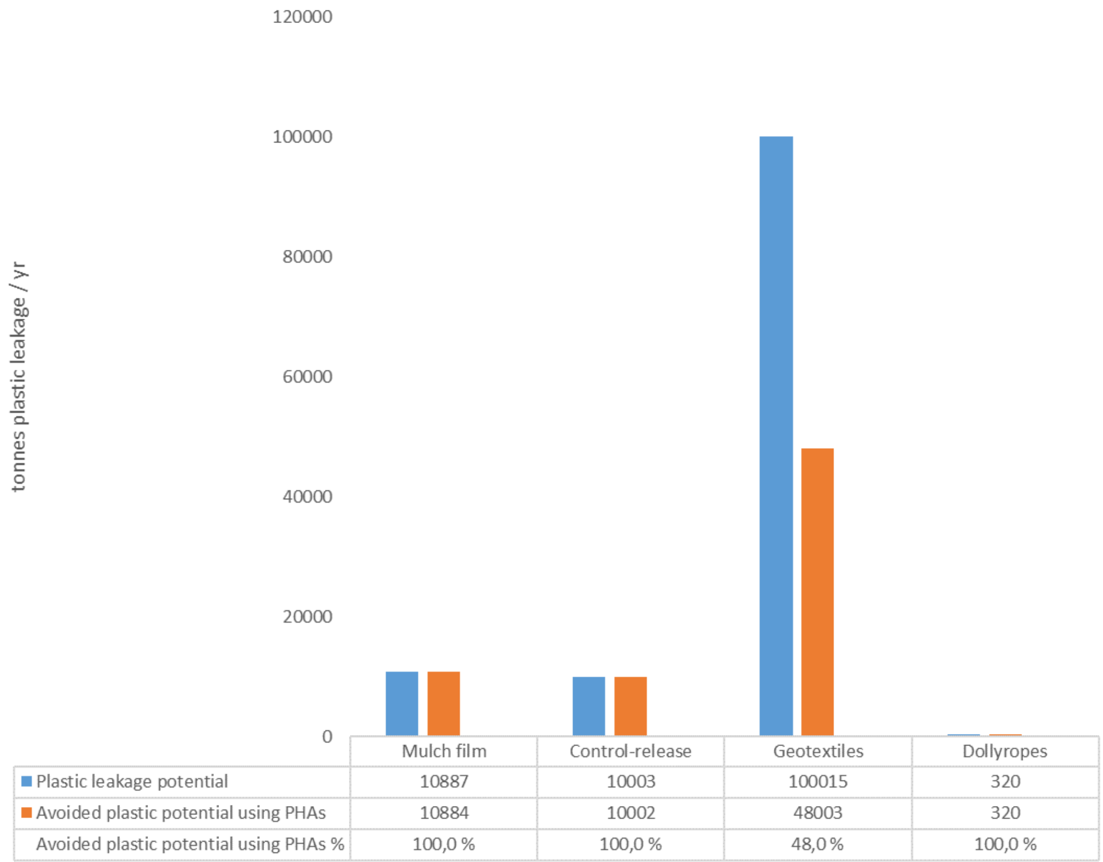


Figure 7: Macroplastic leakage potential associated with annual consumption of product groups in EU, 2021

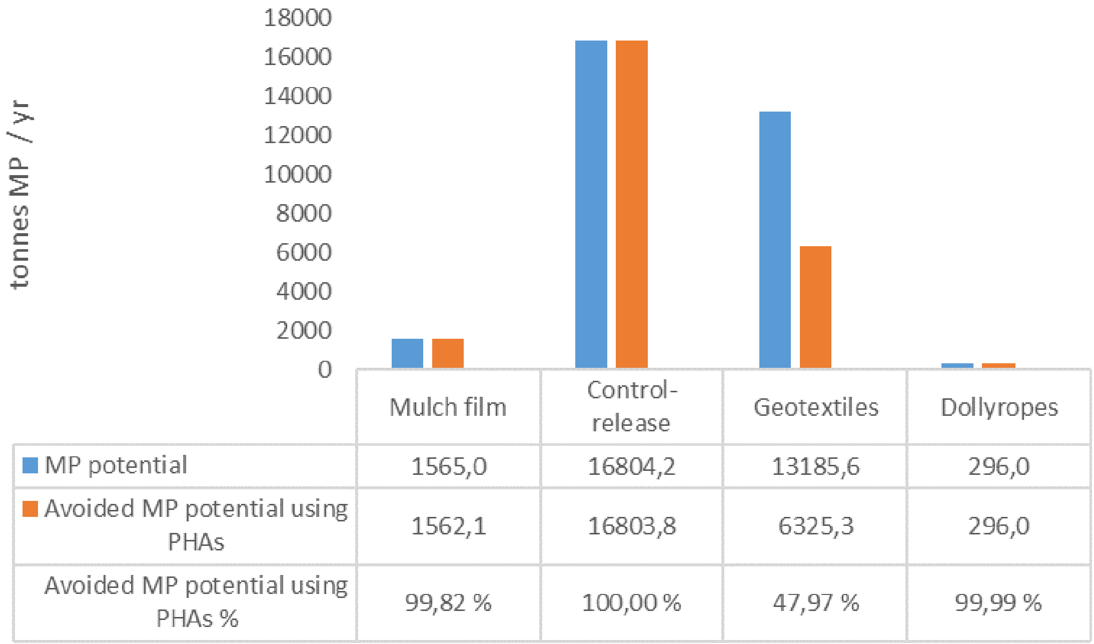
The use of PHA for the investigated product groups shows promising results for mitigating the release of plastics to nature. For mulchfilm, control-release fertilizer and dolly ropes almost all of the currently estimated plastic leakage potential can be avoided using PHA’s (Figure 8). It is only tire wear that limits the PHA’s plastic leak mitigation potential to slightly less than 100% for mulch film, control-release fertilizer and dolly ropes. For geotextiles, the plastic leakage mitigation potential by using PHA instead of fossil polymers is reduced because of a limit to how much of the geotextiles can be assumed to be replaceable with biodegradable materials (see section 2.4.2). Despite the relatively low replacement rate of fossil polymers for biodegradable materials in geotextiles, it is still the product group with the largest potential for mitigation of plastic leakage to the environment (Figure 8). There is, however, a large uncertainty associated with the environmental availability of geotextile plastics, as they are dug into the ground unlike the other products considered in this study that are either leaked on the soil surface, or directly in the marine compartments. The second largest plastic leakage potential stems from mulch film, followed by control-release fertilizer. Figure 9 and Figure 10 show the results of the MFA calculations for net emissions of MP and macroplastic and mitigation potential if PHA’s are used.



	Mulch film	Control-release	Geotextiles	Dollyropes
■ Plastic leakage potential	10887	10003	100015	320
■ Avoided plastic potential using PHAs	10884	10002	48003	320
Avoided plastic potential using PHAs %	100,0 %	100,0 %	48,0 %	100,0 %

Figure 8: Net plastic leakage potential of fossil-based materials compared to PHA mitigation potential of plastic leakage in general.

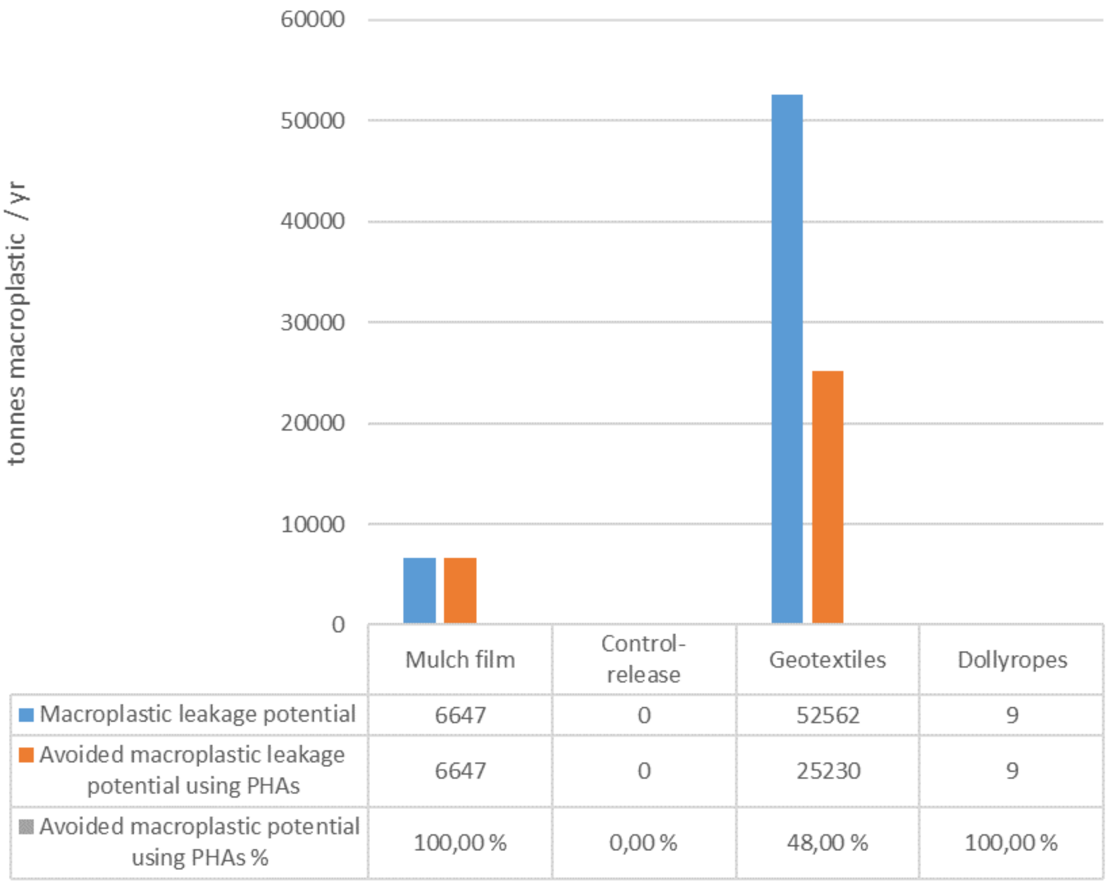
The breakdown of the plastic leakage potentials to MPs demonstrate that the MP potential is greatest for geotextiles, followed by mulch film and control-release fertilizer film (Figure 9). The results show that PHA has the potential to mitigate about 100% of the macroplastic leakage associated with mulch films, control-release fertilizer and dolly ropes. However, there is a small fraction of microplastics emissions that are also caused by tire wear during the transport of materials and products, also the PHA products. This means that PHA products are not fully free from MP leakage (Figure 9), but also note that the MP mitigation potential is not significantly influenced by tire wear. It should also be noted that it can be argued that PHA will have MP leakage for a short time scale, while the biodegradation process occurs (see section 2.2). For geotextiles it is the theoretical replacement rate of PHAs within the product group that limits MP mitigation associated with PHA use.



	Mulch film	Control-release	Geotextiles	Dollyropes
🟦 MP potential	1565,0	16804,2	13185,6	296,0
🟧 Avoided MP potential using PHAs	1562,1	16803,8	6325,3	296,0
Avoided MP potential using PHAs %	99,82 %	100,00 %	47,97 %	99,99 %

Figure 9: Tonnes MP leakage potential of fossil-based materials and avoided MP potential by using PHA biodegradable materials.

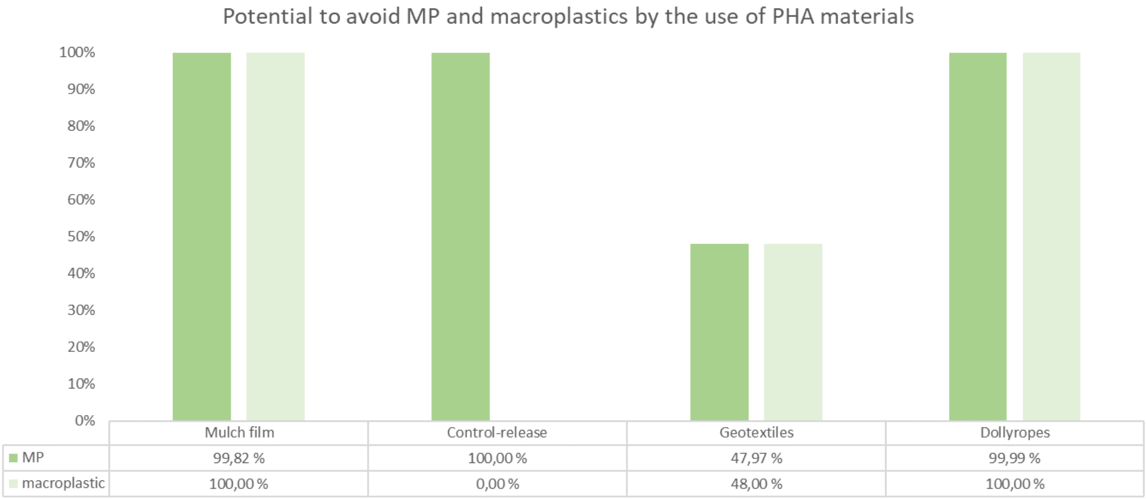
The macroplastic leakage differs in sources and total mass flows compared to MP; this can be seen when comparing Figure 6 and Figure 7. One significant difference between MP and macroplastic potential is for the product groups, dolly ropes and control-release fertilizer. In this study it is assumed that all of the wear and tear of dolly ropes produces MP and all of the control-release fertilizer capsules break apart into MP. This is why dolly ropes and control-release fertilizer are the only product categories that cause more MPs than macroplastic. For the other assessed product groups the net macroplastic leakage potential is significantly higher than for MP (Figure 9 and Figure 10). The mitigation potential of using PHA is greatest in terms of total mass of macroplastics compared to MPs for mulch film and geotextiles and highest for MPs for control-release fertilizer and dolly ropes.



	Mulch film	Control-release	Geotextiles	Dollyropes
🟦 Macroplastic leakage potential	6647	0	52562	9
🟧 Avoided macroplastic leakage potential using PHAs	6647	0	25230	9
⬜ Avoided macroplastic potential using PHAs %	100,00 %	0,00 %	48,00 %	100,00 %

Figure 10: Tonnes macroplastic leakage potential of fossil-based materials and avoided macroplastic potential if using biodegradable PHA materials.

In summary, the relative mitigation potential of plastic leakage in general, MPs and macroplastics from mulch film and control-released fertilizer is very close to 100% (Figure 11). This means that the ecosystem and human health hazards associated with the release of fossil plastic MPs to nature could be avoided using PHA based materials.



	Mulch film	Control-release	Geotextiles	Dollyropes
🟩 MP	99,82 %	100,00 %	47,97 %	99,99 %
🟩 macroplastic	100,00 %	0,00 %	48,00 %	100,00 %

Figure 11: Relative avoided fossil plastic, MPs and microplastics leakage by using PHAs.

5.4 Discussion

The MFA results demonstrate that PHA based materials cause less MP and no macroplastic leakage compared to the fossil polymer materials commonly used today. These results are based on the reported degradability of PHA in the environment. When assessing the life cycle environmental impacts of a product system, material timeframes of 20, 100 to 500 years are common. Therefore, we have excluded potential PHA macro material leakage associated to the environment which degrade within 0.1 to 5.4 years in marine environments. This being said, there is still a risk that PHA poses a short-term macro plastic leakage hazard within the biosphere, causing entanglement during this time. Further research and development of LCIA characterization factors is therefore needed to better predict the potential damage associated with microplastic emissions for PHA based materials. Similarly, as this study focused solely on the fossil plastic to nature caused by traditional materials and PHA mitigation potential of these emissions, this study also neglects the potential negative impact PHA can have on soil chemistry and nutrient availability in agricultural soils. The inhibition of nutrients demonstrates reduced crop yields of corn, which in turn might lead to a higher demand for nutrients or land area. The use of more nutrients would mean that there is a need for more nutrient packaging which may lead to an increased indirect fossil plastic leakage potential. This potential increase in nutrient need is not included in our study but could be considered in further work using MFA and LCA methods in combination to develop better models and estimates for plastic leakage potential of traditional polymers and mitigation potential using PHA.

There are high uncertainties reported in key literature sources regarding the estimated plastic leakage potential associated with fossil plastic polymers, and thus the mitigation potential associated with using PHAs. Siegfried et al. (2017) reported that the terrain, biota, population, waste management systems and proximity to coastal areas all are key factors in determining the MP and macroplastic leakage and runoff to the marine compartment. Kawecki & Nowack (2019) demonstrate high uncertainties in their estimated flows of MPs and macroplastics from the technosphere to and between environmental compartments. Therefore, the MFA results for MP and macroplastics estimated in this study should be similarly regarded as highly uncertain. There is also no consensus on a method to properly account for the multiple times biota can be exposed to fossil MP and macroplastics within each environmental compartment. In this study we opted for double counting the mass of MP and macroplastics transported between the environmental compartments, which is another source of uncertainty associated with the findings. Despite these uncertainties, it is likely that our findings regarding the MP and macroplastic mitigation potential of using PHA instead of fossil plastic polymers are valid.

6 Simplified Impact Assessment

During the initial stages of this work, it was expected that an impact assessment could be performed based on preliminary unpublished impact categories that are currently under development. The CFs needed to carry out an impact assessment are currently only developed for MP and macroplastic in the marine compartment. In this study land based macroplastics are the primary plastic leakage to the environment. Therefore, without a CF for MP and macroplastic on land the impact assessment would have been very skewed, demonstrating only a small fraction of the plastic leakage impact caused by the annual use of the chosen product groups made of petroleum-based polymers. However, many studies point out that there are significant negative impacts on biota and humans associated with MP exposure and biota exposure to macroplastics. Therefore, it is possible without any quantitative damage assessment on specific species to conclude that the impact of MPs and macroplastics are of great concern and reduction in emissions of these plastic leakages is beneficial. In mitigating these forms of pollution, one also omits the associated negative environmental and human impacts. An additional finding through the literature study was that recent literature sources also disclose some negative environmental implications associated with the use and leakage of PHA to nature. However, for PHA leakage this seems to be limited to an ability to inhibit nutrient uptake (Brown et al. 2023). Going forward it would be of interest to examine whether similar nutrient uptake studies had been performed for relevant fossil-based plastic alternatives; also, comparison of the magnitude of these effects when compared to entanglement effects from microplastic releases, which are not likely to occur for PHA products.

The United Nations Environment Programme’s Life Cycle initiative project GLAM (Global Guidance for Life Cycle Impact Assessment Indicators and Methods (GLAM 2021) will publish CFs for MP and microplastic to the marine compartment in 2023. There are researchers internationally working on polymer specific CFs for several polymers in the marine compartment (MarILCA 2022) and there is emerging work for the air and soil compartments, such that impact assessment calculations suitable for inclusion in the LCIA phase of an LCA can be performed in the not-too-distant future.

7 Conclusion

Available literature has partially assessed the exposure and adverse effects of MPs and macroplastics on a variety of species. The body of work on these issues is increasing and expanding at a rapid rate. The scientific consensus is that both MPs and macroplastics have severe impacts on biota, and a growing number of studies are also investigating the potential for MPs being a health hazard for humans. What is known is that MPs are almost everywhere on the globe, both in populated and remote areas, and are found present in all humans across the world. Therefore, using materials that have the potential to mitigate fossil plastic emissions to nature is likely to prove beneficial for ecosystems and humans.

This preliminary study demonstrates that by using PHA, particularly PHBV, plastic leakage to the assessed environmental compartments land, waterways and ocean can be can significantly mitigated. However, there are a growing number of studies that also address possible tradeoffs associated with using PHA. One of these tradeoffs is an inhibition of nutrient uptake, particularly nitrogen and to some extent phosphorus, which reduce crop yields (Brown et al. 2023). Assessing the environmental consequences of tradeoffs is important as it can lead to problem-shifting (i.e. trading one problem for another). However, within the scope of this study the MFA demonstrates that there is a significant MP and microplastic leakage mitigation potential by PHA materials are used instead of traditional fossil-based polymers. Therefore, PHAs will eliminate the associated problems associated to fossil MPs and microplastic found in the literature today. With the emergence of coming CF factors this study provide the mass (M) in the impact assessment equation. As such The MFA results from this study is intended to be used to estimate the impact of the fossil and PHA based impacts once the appropriate CF factors are published and publicly available.

Acknowledgement

NORSUS thanks Invest NL and the PHA2USE Consortium for commissioning and funding this study.

References

Akanyange, S. N., Lyu, X., Zhao, X., Li, X., Zhang, Y., Crittenden, J. C., Anning, C., Chen, T., Jiang, T., & Zhao, H. (2021). Does microplastic really represent a threat? A review of the atmospheric contamination sources and potential impacts. Science of The Total Environment, 777, 146020. https://doi.org/10.1016/j.scitotenv.2021.146020

Altaee, N., El-Hiti, G. A., Fahdil, A., Sudesh, K. and Yousif, E. (2016). Biodegradation of different formulations of polyhydroxybutyrate films in soil. SpringerPlus (2016) 5:762. DOI 10.1186/s40064-016-2480-2

Arcos-Hernandez, M. V., Laycock, B., Pratt, S., Donose, B. C., Nikolić, M. A. L., Luckman, P., Werker, A., & Lant, P. A. (2012). Biodegradation in a soil environment of activated sludge derived polyhydroxyalkanoate (PHBV). Polymer Degradation and Stability, 97(11), 2301–2312. https://doi.org/10.1016/j.polymdegradstab.2012.07.035

Askham, C., Pauna, V. H., Boulay, A.-M., Fantke, P., Jolliet, O., Lavoie, J., Booth, A. M., Coutris, C., Verones, F., Weber, M., Vijver, M. G., Lusher, A., & Hajjar, C. (2023). Generating environmental sampling and testing data for micro- and nanoplastics for use in life cycle impact assessment. Science of The Total Environment, 859, 160038. https://doi.org/10.1016/j.scitotenv.2022.160038

Bauchmüller, V., Carus, M., Chinthapalli, R., Dammer, L., Hark, N., Partanen, A., Ruiz, P., & Lajewski, S. (2021). Biosinn - Products for which Biodegradation makes sense. nova-Institut für politische und ökologische. https://renewable-carbon.eu/publications/product/biosinn-products-for-which- biodegradation-makes-sense-pdf/

Boyer, J., Rubalcava, K., Booth, S., & Townsend, H. (2022). Proof-of-concept model for exploring the impacts of microplastics accumulation in the Maryland coastal bays ecosystem. Ecological Modelling, 464, 109849. https://doi.org/10.1016/j.ecolmodel.2021.109849.

Brown, R. W., Chadwich, D. R., Zang, H., Graf, M., Liu, X., Wang, K., Greenfield, L. M. and Jones, Davey, L. Bioplastic (PHBV) addition to soil alters microbial community structure and negatively affects plant-microbial metabolic functioning in maize. (2023). Journal of Hazardous Materials, 441, 1s29959. https://doi.org/10.1016/j.jhazmat.2022.129959.

Brunner, P. H. and Rechberger, H. 2004. Practical handbook of material flow analysis. The international journal of Life cycle assessment, 9, 337-338.

Carpenter, E.J. & Smith, K., (1972). Plastics on the sargasso sea surface. Science 175 (4027), 1240–1241.

Carpenter, E.J., Anderson, S.J., Harvey, G.R., Miklas, H.P., Peck, B.B. (1972). Polystyrene spherules in coastal waters. Science 178 (4062), 749–750.

Corella-Puertas, E., Guieu, P., Aufoujal, A., Bulle, C., & Boulay, A. M. (2022). Development of simplified characterization factors for the assessment of expanded polystyrene and tire wear microplastic emissions applied in a food container life cycle assessment. Journal of Industrial Ecology, 26(6), 1882–1894. https://doi.org/10.1111/jiec.13269

Cowger, W., Booth, A. M., Hamilton, B. M., Thaysen, C., Primpke, S., Munno, K., Lusher, A.L., Dehaut, A., Vaz, V.P., Liboriron, M., Devriese, L.I., Hermabessiere, L., Rochman, C., Athey, S.N., Lynch, J.M., De Frond, H., Gray, A., Jones, O.A.H., Brander, S., Steele, C., Moore, S., Sanchez, A., Nel, H. (2020). Reporting Guidelines to Increase the Reproducibility and Comparability of Research on Microplastics. Applied Spectroscopy. https://doi.org/10.1177/0003702820930292

Croxatto Vega, G., Gross, A., & Birkved, M. (2021). The impacts of plastic products on air pollution - A simulation study for advanced life cycle inventories of plastics covering secondary microplastic production. Sustainable Production and Consumption, 28, 848–865. https://doi.org/10.1016/j.spc.2021.07.008.

Dilkes-Hoffman, L. S., Lant, P. A., Laycock, B. and Pratt, S. (2019). The rate of biodegradation of PHA bioplastics in the marine environment: A meta-study. Marine Pollution Bulletin 142 (2019) 15–24. https://doi.org/10.1016/j.marpolbul.2019.03.020.

Elia, V., M.G. Gnoni, and F. Tornese, Measuring circular economy strategies through index methods: A critical analysis. Journal of Cleaner Production, 2017. 142: p. 2741-2751.

FAO. 2021. Assessment of agricultural plastics and their sustainability. A call for action. Rome. https://doi.org/10.4060/cb7856en

Frias, J. P. G. L., & Nash, R. (2019). Microplastics: Finding a consensus on the definition. Marine Pollution Bulletin, 138, 145–147. https://doi.org/10.1016/j.marpolbul.2018.11.022.

GESAMP: Sources, Fate and Effects of Microplastics in the Marine Environment: Part 2 of a Global Assessment, 93rd edn. (2016).

Global guidance for life cycle impact assessment indicators and methods (GLAM) - life cycle initiative. Life Cycle Initiative. (2021, February 7). Retrieved January 20, 2023, from https://www.lifecycleinitiative.org/activities/key-programme-areas/life-cycle-knowledge-consensus-and-platform/global-guidance-for-life-cycle-impact-assessment-indicators-and-methods-glam/

Gontard, N., David, G., Guilbert, A., & Sohn, J. (2022). Recognizing the long-term impacts of plastic particles for preventing distortion in decision-making. Nature Sustainability, 5(6), 472–478. https://doi.org/10.1038/s41893-022-00863-2

Greene, J. P. (2021). 7 - Commodity Plastics. In: GREENE, J. P. (ed.) Automotive Plastics and Composites. William Andrew Publishing.

Han, N., Zhao, Q., Ao, H., Hu, H., & Wu, C. (2022). Horizontal transport of macro- and microplastics on soil surface by rainfall induced surface runoff as affected by vegetations. Science of The Total Environment, 831, 154989. https://doi.org/10.1016/j.scitotenv.2022.154989

Hasan, R. (2020). An overview of geotextiles: Industrial application in technical textiles. Journal of Textile Science & Fashion Technology, 4(4). https://doi.org/10.33552/jtsft.2020.04.000593

Junaid, M., Liu, S., Chen, G., Liao, H., & Wang, J. (2023). Transgenerational impacts of micro(nano)plastics in the aquatic and terrestrial environment. Journal of Hazardous Materials, 443, 130274. https://doi.org/10.1016/j.jhazmat.2022.130274

Kawecki and Nowack (2019). Polymer-Specific Modelling of the Environmental Emissions of Seven Commodity Plastics As Macro- and Microplastics. Environ. Sci. Technol. 2019, 53, 9664−9676. DOI: 10.1021/acs.est.9b02900.

Klöpffer, W. & Grahl, B. (2014). Life cycle assessment (Lca) A guide to best practise. Wiley.

Lavoie, J., Boulay, A. M., & Bulle, C. (2021). Aquatic Micro- and Nano-plastics in life cycle assessment: Development of an effect factor for the quantification of their physical impact on biota. Journal of Industrial Ecology, 26(6), 2123–2135. https://doi.org/10.1111/jiec.13140

Lusher, A. L., Munno, K., Hermabessiere, L., & Carr, S. (2020). Isolation and extraction of microplastics from environmental samples: An evaluation of practical approaches and recommendations for further harmonization. Applied Spectroscopy, 74(9), 1049–1065. https://doi.org/10.1177/0003702820938993

Lusher, A. L., Hurley, R., Arp, H. P. H., Booth, A. M., Bråte, I. L. N., Gabrielsen, G.W., Gomiero, A., Gomes, T., Grøsvik, B. E., Green, N., Haave, M., Hallanger, I. G., Halsband, C., Herzke, D., Joner, E. J., Kögel, T., Rakkestad, K., Ranneklev, S. B., Wagner, M., & Olsen, M. (2021). Moving forward in microplastic research: A Norwegian perspective. Environment International, 157, 106794. https://doi.org/10.1016/j.envint.2021.106794

Marine impacts in life cycle assessment. MariLCA. (2022, November 21). Retrieved January 17, 2023, from https://marilca.org/

Mas-Castella. J., Urmenteta, J., Lafuente, R., Navarrete, A. and Guerrero, R. (1995). Biodegradation of *poly- β-hydroxualkanoates in Anaerobic Sediments*. International Biodeterioration & Biodegradation (1995) 155-114.

The Online Materials Information Resource. MatWeb. (n.d.). Retrieved January 20, 2023, from https://www.matweb.com/search/QuickText.aspx

Maga, D., Vazquez-Rowe, I., Verones, F., Boulay, A.-M., Corella-Puertas, E., & Askham, C. (n.d.). Consideration of plastic emissions in life cycle assessments. In Springer Handbook of Circular Plastics Economy (pp. 2–37). book chapter, Springer. *in press*.

Mo, A., Zhang, Y., Gao, W., Jiang, J. and He, D. (2023): Environmental fate and impacts of biodegradable plastics in agricultural soil ecosystems. Applied Soil Ecology 181 (2023) 104667. https://doi.org/10.1016/j.apsoil.2022.104667

OECD. (2021). Policies to reduce microplastics pollution in water: Focus on textiles and tyres. OECD. Retrieved January 15, 2023, from https://www.oecd.org/environment/policies-to-reduce-microplastics-pollution-in-water-7ec7e5ef-en.htm

Ong Su Yean, Chee Jiun Yee, Sudesh Kumar. Degradation of polyhydroxyalkanoate (PHA): a review. J. Sib. Fed. Univ. Biol. (2017). 10(2), 211-225. DOI: 10.17516/1997-1389-0024.

Pauna, V. H., & Askham, C. (2022). Using information flow analysis to establish key data gaps in the assessment of marine microplastic pollution. Journal of Industrial Ecology, 26(6), 1895–1907. https://doi.org/10.1111/jiec.13312

Peano, L., Kounina, A., Magaud, V., Chalumeau, S., Zgola, M., & Boucher, J. (2020). Plastic leak project methodological guidelines. https://quantis-intl.com/report/the-plastic-leak-project-guidelines/

Prambauer, M., Wendeler, C., Weitzenböck, J., & Burgstaller, C. (2019). Biodegradable geotextiles – an overview of existing and potential materials. Geotextiles and Geomembranes, 47(1), 48–59. https://doi.org/10.1016/j.geotexmem.2018.09.006

Rai, P. K., Lee, J., Brown, R. J. C., & Kim, K.-H. (2021). Environmental fate, ecotoxicity biomarkers, and potential health effects of micro- and nano-scale plastic contamination. Journal of Hazardous Materials, 403, 123910. https://doi.org/10.1016/j.jhazmat.2020.123910

Ravanbakhsh, M., Ravanbakhsh, M., Jamali, H. A., Ranjbaran, M., Shahsavari, S., & Jaafarzadeh Haghighi Fard, N. (2022). The effects of storage time and sunlight on microplastic pollution in bottled mineral water. Water and Environment Journal. https://doi.org/10.1111/wej.12829.

Saling, P., Gyuzeleva, L., Wittstock, K., Wessolowski, V., & Griesshammer, R. (2020). Life cycle impact assessment of microplastics as one component of marine plastic debris. The International Journal of Life Cycle Assessment, 25(10), 2008–2026. https://doi.org/10.1007/s11367-020-01802-z

ScienceDirect.com | Science, health and medical journals, full text articles and books. (n.d.). Retrieved January 20, 2023, from https://www.sciencedirect.com/

Siegfried, M., Koelmans, A. A., Besseling, E. and Kroeze, C. (2017), Export of microplastics from land to sea. A modelling approach. Water Research 127 (2017). 249e257.https://doi.org/10.1016/j.watres.2017.10.011

Sundt, P., Schulze, P.-E., & Syversen, F. (2014). Sources of microplastic- pollution to the marine environment Project report.

van Eck, N. J., & Waltman, L. (2022). VOSViewerVersion (1.6.18). VOSViewer visualizing scientific landscapes. Universiteit Leiden & CWTS. https://www.vosviewer.com/

Veiga, J.M., Fleet, D., Kinsey, S., Nilsson, P., Vlachogianni, T., Werner, S., Galgani, F., Thompson, R.C., Dagevos, J., Gago, J., Sobral, P. and Cronin, R.; 2016; Identifying Sources of Marine Litter. MSFD GES TG Marine Litter Thematic Report; JRC Technical Report; EUR 28309; doi:10.2788/018068

Verschoor, A., de Poorter, L., Dröge, R., Kuenen, J., & de Valk, E. (2016). (RIVM Report 2016-0026). Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear (pp. 1–73). Bilthoven, Netherlands: National Institute for Public Health and the Environment.

Wang, Y., Zhou, B., Chen, H., Yuan, R., & Wang, F. (2022). Distribution, biological effects and biofilms of microplastics in freshwater systems - a review. Chemosphere, 299, 134370. https://doi.org/10.1016/j.chemosphere.2022.134370

Woods, J.S., Rødder, G., Verones, F.: An effect factor approach for quantifying the entanglement impact on marine species of macroplastic debris within life cycle impact assessment Ecological Indicators 99, 61–66 (2019). doi: 10.1016/j.ecolind.2018.12.018.

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

Appendix 1: All Keywords and Thesaurus

Will be made available after publication of a scientific article

Appendix 2: Full Keyword Co-Occurrence Analysis

Will be made available after publication of a scientific article

Appendix 3: MFA Transfer Coefficients

Activities and Coefficients						
	Mulch film	Control-release Fertilizer	Geotextiles	Dolly ropes	PHAs	Unit MFA
A1 - Crude oil to plastic	1,00	1,00	1,00	1,00		tonnes
k1,2						
k1,0 - Plastic to nature						
k1,0 SD						
T1-2 km	250	250	250	250	250	km
T1,2	7,56E-06	7,56E-06	7,56E-06	7,56E-06	3,02E-05	tonnes
A2 - Production	1,00	1,00	1,00	1,00	4,00	tonnes
k2,3						
k2,0 - Plastic to nature	0,0001	0,0001	0,0001	0,0001	0,0001	
k2,0 - SD						
T2-3 km	500	500	500	500	500	km
T2,3	1,51E-05	1,51E-05	1,51E-05	1,51E-05	6,05E-05	tonnes
A3 - Manufacturing production	1	1	1	1	4	tonnes
k3,4						

k3,0 - Plastic to nature						
k3,0 SD						
T3-4 km	500	500	500	500	500	km
T3,4	1,51E-05	1,51E-05	1,51E-05	1,51E-05	6,05E-05	tonnes
A4 - Use	1	1	1	1	4	tonnes
k4,5	0,90	0,00	0,00	0,72	1,00	
k4,0 - Plastic to nature	0,10	1,00	1,00	0,28	0,00	
k4,0 - SD				0,16	0	
T4-5 km	50			50	50	km
T4,5	1,36E-06	0,00E+00	0,00E+00	1,08E-06	6,05E-06	tonnes
A5 - Waste management	0,90	0,00	0,00	0,72	4,00	tonnes
k5,6 To landfill	0,37			0,37	0	
k5,Incinerated	0,44			0,44	0	
k5,7 To recycling	0,14			0,14	0	
k5,0 - Mismanaged waste	0,05			0,05	1	
k5,0 - SD						
T5-6 km	50			50	50	km
T5,6	5,03E-07			4,01E-07	0	tonnes
T5-7 km	170			170	170	km
T5,7	6,48E-07			5,16E-07	0	tonnes
A6 - Landfill	0,33	0,00	0,00	0,27	0,00	tonnes
Δ6	0,33			0,27	0,00	
k6,0 - Plastic to nature						
k6,0 - SD						
A7 - Recycling	0,13	0,00	0,00	0,10	0,00	tonnes
k7,2						
k7,0 - Plastic to nature	0,00			0,00	0,00	
k7,0 - SD						
T7-2 km	500	500	500	500	500	km
T7,2	1,51E-05	1,51E-05	1,51E-05	1,51E-05	1,51E-05	tonnes

Norwegian Institute for
Sustainability Research

"Mark and insert co authors logo"