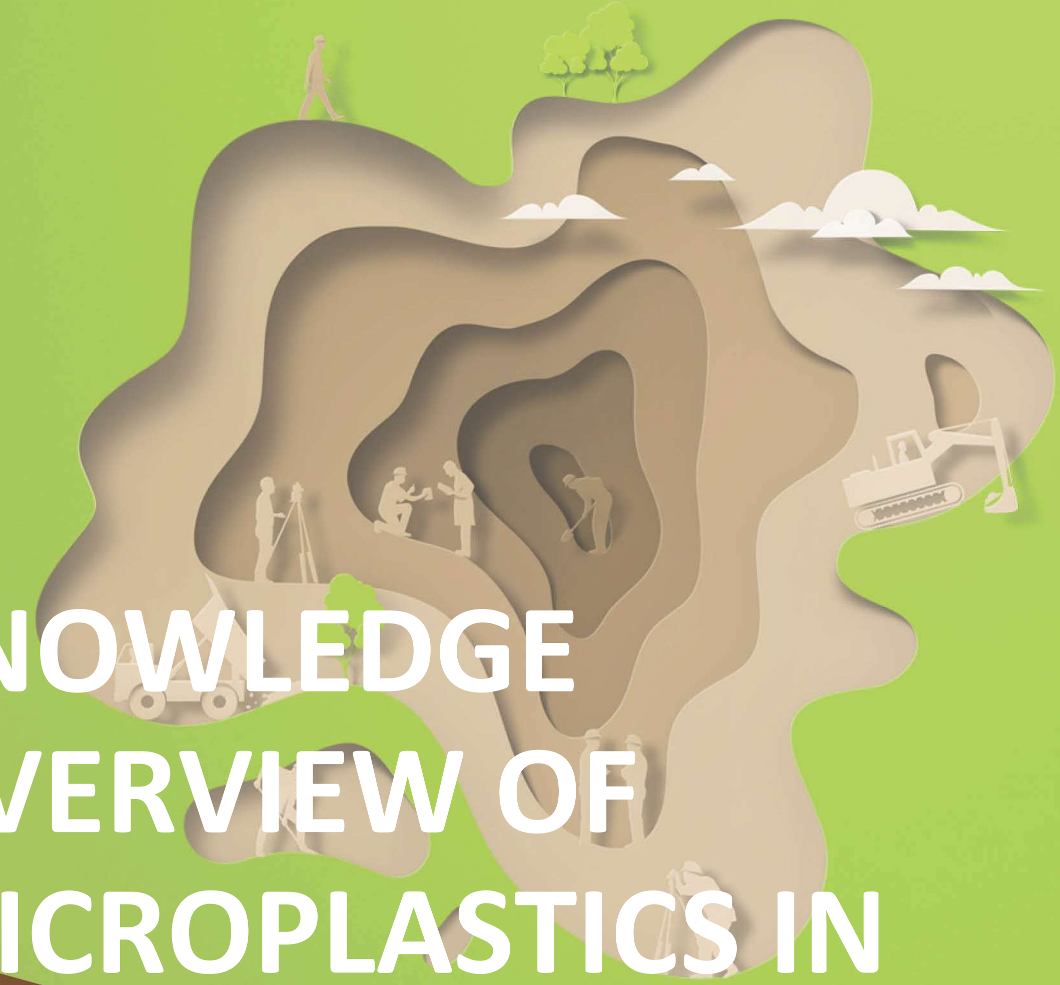

KNOWLEDGE OVERVIEW OF MICROPLASTICS IN SOIL, GROUNDWATER AND SEDIMENTS

Flanders
State of the Art

WE MAKE TOMORROW BEAUTIFUL
OVAM

OVAM.VLAANDEREN.BE

/ Report

KNOWLEDGE OVERVIEW OF MICROPLASTICS IN SOIL, GROUNDWATER AND SEDIMENTS

Publication date / 2.06.2025

DOCUMENT DESCRIPTION

1 *Title of publication:*
Knowledge overview of microplastics in soil, groundwater and sediments

2 *Responsible Publisher:*
OVAM

3 *Legal deposit number:* D/2025/5024/22

4 *Key words:*
Microplastics, soil, sediment, groundwater

5 *Summary:*
This report provides an in-depth analysis of microplastic pollution in soils, sediments and groundwater. The report consists of three parts: (A) a comprehensive overview of the sources, concentrations, risks, measurement techniques, regulations and remediations strategies of microplastics. (B) an ecotoxicological and human health evaluation based on part A and (C) future policy studies and policy-supporting research.

6 *Number of pages: 121*

7 *Number of tables and figures:* 21 figures and 19 tables (including supplementary material)

8 *Date of publication:*02-06-2025

9 *Price*: /*

10 *Steering committee and/or author:*
Lisa Joos, Nick Krekelbergh, Yin Liu, Benthe Van Buyten, Jozefien Demeulenaere, Patria Novita Kusumawardani, Stefaan De Neve, Caroline De Tender (UGent)

11 *Contact person(s):*
Laetitia Six, Griet Van Gestel (OVAM)

Other publications about the same subject:

Part of the researchers involved in this study are funded by the European Research Council under project 101075944, or the "MiCoS project" (https://www.micos.ugent.be).

INDEX

EXECUTIVE SUMMARY

Microplastics, all plastic polymer particles smaller than 5 mm, are ubiquitous in the environment globally. Regulations regarding microplastics input and presence in soil ecosystems are nevertheless absent, majorly due to the lack of information regarding their presence and effects on the environment and humans. In the last five years, significant efforts have been made to improve methodologies to measure microplastics and studies have been performed related to toxicological effects on soil, fauna and flora and humans to address these issues. This report, commissioned and supervised by OVAM, the Public Waste Agency in the region of Flanders (Belgium), summarizes current insights and offers recommendations to guide future policy actions.

During the second Agrifoodplast conference held in Brussels in April 2025, four key messages were addressed to policy makers representing the European Union by us, researchers working on plastic pollution. First, plastic pollution is ubiquitous in European soils. Second, this pollution comes from different sources. Third, plastic pollution is non-reversible and will only increase in the future even if we stop producing plastic now. Fourth, nanoplastics can be taken up by plants and translocated to edible parts. These key messages are a good representation of the content provided here, summarized in three parts.

In **PART A**, we give an overview of the current knowledge related to microplastic pollution, with the focus on soil ecosystems, and to some extent also groundwater and sediment. This part not only addresses the current detected concentrations of microplastics and their potential sources, but also the measurement techniques, known risks for the environment and humans, the behavior of microplastics in soils and sediments, current regulations and soil remediation. **PART B** presents an ecotoxicological and human health evaluation, building on the findings of PART A. It integrates available data to assess the potential impacts of microplastics on terrestrial organisms and human health. **PART C** focuses on future policy development and identifies key research needs to support effective policymaking. It outlines recommendations for improving risk assessments and highlights knowledge gaps that must be addressed to guide regulatory actions and sustainable solutions. Underneath an extensive summary can be found for those three parts:

PART A | Knowledge overview

Chapter 1 provides an introduction to micro- and nanoplastics. Microplastics (MP) are defined as plastic particles smaller than 5 mm, while nanoplastics (NP) typically range between 1 nm and 1 µm. Microplastics are further categorized into primary microplastics, which are manufactured as small sized particles for specific uses (e.g. microbeads in cosmetics), and secondary microplastics, which result from the fragmentation of larger plastic products through environmental processes like weathering. The most commonly produced plastic polymers contributing to microplastic pollution include polypropylene (PP), polyvinyl chloride (PVC), and various forms of polyethylene (PE), such as LDPE (low density PE), LLDPE (linear LDPE), and HDPE (high density PE). In addition to the base polymers, plastics often contain a range of additives—intentionally added chemical compounds designed to enhance performance. Many of these additives are considered chemicals of concern due to their persistence, bioaccumulation potential, mobility in the environment, and toxicity (PBMT). Microplastics are highly diverse in structure and behaviour. These particles vary widely in shape—including fibers, fragments, beads, spheres, and pellets—and in composition, as they can consist of multiple polymers, additives, and sizes.

This complex and variable nature makes microplastics a multifaceted environmental issue, requiring careful study of their characteristics and sources.

In **Chapter 2**, the ubiquity of microplastics is highlighted, as they are found in air (8%), water (12%), and soil (80%). Recent research by RIVM (The Netherlands), identifies the most significant sources of microplastic pollution as: tyre wear particles (12 500 ton), plastic granules used in manufacturing (primary microplastics; 9 300 ton), and plastic waste that degrades into secondary microplastics (8 600 ton) (Quik et al., 2024). Other, smaller sources include fibers from textiles (3 100 ton), intentionally produced microplastics (2 000 ton), paints and coatings (900 ton) and agriculture (880 ton). While agriculture is a relatively minor contributor overall, it is of particular importance for our food system. Here, microplastics can originate from compost, sewage sludge, controlled-release fertilizers (CRFs), mulching, and irrigation activities. Once in the soil, microplastics can migrate both horizontally and vertically, raising concerns about their potential entry into the food chain and broader ecosystems.

In **Chapter 3**, data of multiple studies on microplastic concentrations in soil, groundwater, and sediments were collected and summarized. The soil database includes 85 studies published between 2018 and 2024, encompassing 199 soil samples across various regions, with a primary focus on Asia (57%) and Europe (27%). The studies mainly target agricultural soils (56%), with smaller representations of forest, grassland, greenhouse, wetland, and urban soils. The results reveal considerable variation in MP concentrations, with 51% of studies reporting less than 1 000 MP particles per kg of soil, 37% between 1 000 and 10 000 particles per kg soil, and 12% reporting concentrations exceeding 10 000 particles per kg soil. In groundwater, the concentration of microplastics, based on 58 studies from 25 countries, ranges from 0 to 6 832 particles per liter. Groundwater, being a vital resource for human consumption and the environment, presents a significant concern for microplastic contamination, as it can affect drinking water sources globally. For sediments, the MP-SED database was used including data from studies between 2004 – 2023. Research on microplastics in sediments has been conducted primarily in Asia (32%), followed by Europe (31%) and North America (30%), with various environments studied, such as lakes, rivers, and estuaries. Results show significant variation in concentrations, ranging from 0 to 74 800 MP particles per kg of dry sediment, with the highest concentrations found in areas with industrial activity. Flood events were shown to significantly alter MP concentrations, indicating that environmental factors, such as river connectivity and atmospheric deposition, influence microplastic distribution in sediments across different ecosystems.

This chapter underscores the importance of ongoing studies monitoring microplastics to better understand their widespread presence and impact. While current research has primarily focused on agricultural soils, it is crucial to expand efforts to include soils under other land uses, ensuring a more comprehensive understanding of microplastic contamination across diverse ecosystems.

Chapter 4 outlines various techniques for detecting MPs in soil samples, focusing on both conventional and newly emerging methods. Conventional techniques typically involve a series of steps, including pre-treatment (drying), purification, MP extraction, filtration, identification, and quantification, with common methods like FTIR spectroscopy, Raman spectroscopy, visual inspection, and staining dyes used for analysis. Emerging methods are also discussed, particularly in the areas of purification, extraction, and detection, with a table summarizing the advantages and disadvantages of each technique.

However, to date, there is no consensus on the best analytical methods, and challenges remain, especially when dealing with specific matrices (e.g., those rich in organic matter) or rubber particles, which can complicate the analysis. Within this chapter, a focus is placed on the Quality Assurance and Quality Control (QA/QC), as the quantification of MPs in environmental samples remains a challenging task, with issues of contamination, overestimation, and underestimation affecting results. These challenges are particularly pronounced during the extraction phase, where proper validation studies and the use of blanks are essential to ensure reliable data. The chapter provides a table summarizing the essential QA/QC procedures that help address these issues and improve the accuracy of results.

Chapter 5 explores the effects of MNPs on soil, plant and human health and the concept of safe concentration limits of MPs in different environments. MNPs can alter key soil characteristics such as porosity, bulk density, and water availability. The presence of MNPs can reduce soil porosity and bulk density, affecting soil structure and water retention, which in turn can lead to increased evaporation and reduced permeability. The effects of MNPs on soil organisms are equally concerning as earthworms, nematodes, and arthropods, can ingest MNP particles, leading to reduced growth, survival, and reproduction. Microbial communities change upon MPs presence, and additionally MPs can also create new niches on which microbial biofilms can form. Taken together, impacts on soil functions are expected. In plants, MNPs can have both direct and indirect effects. Directly, MNPs can be absorbed by plant roots and leaves, causing stress and impairing growth. Smaller particles, in particular, are more likely to penetrate plant tissues and cause significant damage. Indirectly, MNPs can disrupt the soil's microbial ecosystem, impairing the beneficial relationships between plants and microorganisms, such as nitrogen-fixing bacteria and mycorrhizal fungi. This interference can reduce nutrient uptake and affect plant growth and productivity.
Regarding human health, Vercauteren et al. (2023a) highlights ingestion as the main exposure route, with inhalation also of concern. MPs have been detected throughout the human body. MPs have been detected in the food chain, including in fish, bottled water, honey, and agricultural products. In addition, the chapter also discusses safe MP concentrations for different environments.

Chapter 6 discusses the role of MNPs in contaminant transport. MNPs can adsorb pollutants and act as vectors, carrying them through ecosystems, potentially making immobile contaminants mobile and allowing them to reach deeper soil layers or groundwater. However, the ability of MP to enhance contaminant transport is limited by factors like desorption rates, soil properties, and water flow conditions.

Chapter 7 focuses on the behaviour of MPs in soils and sediments, highlighting how soil structure influences their distribution and dynamics. MPs can be incorporated into soil aggregates, with larger macro-aggregates holding more particles, while soil texture, MP size, and shape affect their mobility and migration through the soil profile. Migration is mainly driven by bioturbation from soil organisms like earthworms and crop roots, with factors such as soil porosity and wetting-drying cycles influencing their transport, potentially reaching groundwater, although this has not been conclusively proven in field studies.

In **Chapter 8**, existing and emerging regulations regarding MP pollution in environmental compartments beyond soil, such as marine and freshwater systems, are discussed, along with the current regulatory gaps in soil, groundwater, and sediment. While international efforts have been more focused on aquatic environments, terrestrial compartments remain underregulated. The EU leads with the Zero Pollution Action Plan and recent bans on intentionally added MPs, yet most countries only address MPs indirectly, for example through bans on microbeads in cosmetics. Monitoring and regulatory frameworks for MPs in soil are still in early stages, with limited national initiatives such as France's MICROSOF project. Current regulations are fragmented and often lack a lifecycle approach. A global treaty on plastic pollution is under negotiation, aiming to address the full plastic lifecycle and establish binding targets.

The chapter highlights the urgent need for coordinated global action, stronger regulations, and harmonized monitoring standards to address MP pollution across all environmental compartments effectively.

In **Chapter 9**, various remediation techniques for MP in soils are discussed, focusing on their mechanisms, applicability, and limitations. Several techniques exist to remediate MPs in soils, though none offer complete removal, and their effectiveness varies depending on MP type, concentration, soil conditions, and method used. Pyrolysis and photocatalytic degradation are ex-situ methods for topsoil, with pyrolysis offering high MP reduction but damaging soil and requiring high energy, while photocatalysis is less invasive but depth-limited. Magnetic extraction is a gentler ex-situ option with comparable efficiency. In-situ approaches include phytoremediation, which uses specific plants to extract or immobilize MPs but needs long cycles, and microbial degradation, which is cost-effective and eco-friendly but slowed by microbial limitations and uncertain outcomes. Each method has context-specific benefits, with no single solution fitting all conditions.

PART B | Microplastic problems and risks

Understanding the risks of MPs to ecosystems and human health requires more than just measuring how much is present. Key characteristics—such as size, shape, polymer type, chemical additives, and even the presence of biofilms—must also be considered. Experts emphasize the need for a new framework that fully reflects the complex nature of microplastics. To improve future research, studies should use environmentally relevant concentrations and include a mix of microplastic types and forms. It is also important to expand ecotoxicological testing across various soil types and plant species to better reflect real-world conditions.

PART C | Future policy development and key research

Microplastics in soils and groundwater are an emerging concern, but research in this area is still in its early stages. **The first chapter in PART C highlights the major knowledge gaps** that need to be addressed to develop effective monitoring and mitigation strategies. Key unknowns include the true sources of microplastics—especially from everyday consumer goods and biobased plastics—their concentrations in the environment, and how they behave once in soils or sediments. Current methods struggle to detect smaller particles, making it likely that MP pollution is being significantly underreported.

There is also limited understanding of how different types, shapes, and additives in plastics affect ecosystems and human health. Most studies only test individual plastics under simplified conditions, which doesn't reflect real-world exposure to complex plastic mixtures. Furthermore, the environmental and health impacts of thousands of plastic additives remain largely unstudied and unregulated.

In the second chapter of PART C, policy recommendations are explored. A key priority for effective regulation is gaining accurate insight into current microplastic pollution levels in soils, water, air, and food. This data is essential for assessing potential risks to human health, plant development, and the environment.
Despite growing attention, major challenges remain in how microplastics are measured. Current techniques often miss smaller particles and vary significantly between laboratories. To address this, scientists and

policymakers must collaborate to develop standardized methods, launch regional monitoring campaigns, and build on existing European initiatives such as MiCoS, Papillons, and MINAGRIS.

Belgium is in a strong position to take the lead, leveraging existing networks like Cmon and LUCAS to initiate national monitoring. Ideally, soils would be assessed every 3 to 5 years, with a focus on topsoil layers and strict sampling protocols to prevent contamination.

The way microplastic contamination is reported—by particle count or weight—depends on the chosen method and the research objective. Harmonizing these approaches will be essential for producing comparable data and informing future plastic policy at both national and European levels.

SAMENVATTING

Microplastics, alle plastic polymeer deeltjes kleiner dan 5 mm, zijn wereldwijd alomtegenwoordig in het milieu. Regelgeving met betrekking tot de invoer en aanwezigheid van microplastics in bodemsystemen ontbreekt echter, voornamelijk vanwege het gebrek aan informatie over hun aanwezigheid en effecten op mens en milieu. In de afgelopen vijf jaar zijn er aanzienlijke inspanningen geleverd om methodologieën te verbeteren voor het meten van microplastics en zijn er studies uitgevoerd met betrekking tot de toxicologische effecten op bodem, fauna en flora en mensen om deze kwesties aan te pakken. Dit rapport, in opdracht en onder toezicht van OVAM, vat de huidige inzichten samen en biedt aanbevelingen voor toekomstige beleid.

Tijdens de tweede Agrifoodplast-conferentie, gehouden in Brussel in april 2025, werden vier kernboodschappen overgebracht aan Europese beleidsmakers door ons, onderzoekers die werken op plasticvervuiling. Ten eerste is plasticvervuiling alomtegenwoordig in Europese bodems. Ten tweede komt deze vervuiling uit verschillende bronnen. Ten derde is plasticvervuiling onomkeerbaar en zal deze alleen maar toenemen in de toekomst, zelfs als we nu stoppen met het produceren van plastic. Ten vierde kunnen nanoplastics door planten worden opgenomen en worden getransporteerd naar eetbare delen. Deze kernboodschappen zijn een goede weergave van de inhoud die hier wordt verstrekt, samengevat in drie delen.

In **DEEL A** geven we een overzicht van de huidige kennis met betrekking tot microplasticvervuiling, met de focus op bodemsystemen, en tot op zekere hoogte ook grondwater en sediment. Dit deel behandelt niet alleen de huidige gedetecteerde concentraties van microplastics en hun potentiële bronnen, maar ook de meetmethoden, bekende risico's voor mens en milieu, het gedrag van microplastics in bodems en sedimenten, huidige regelgeving en bodemsaneringstechnieken. **DEEL B** presenteert een ecotoxicologische en humane risico-evaluatie, gebaseerd op de bevindingen van DEEL A. Het integreert beschikbare gegevens om de potentiële impact van microplastics op terrestrische organismen en de menselijke gezondheid te beoordelen. **DEEL C** richt zich op toekomstige beleidsontwikkeling en identificeert belangrijke onderzoeksbehoeften om effectief beleid te ondersteunen. Het schetst aanbevelingen voor het verbeteren van risicobeoordelingen en benadrukt kennisleemtes die moeten worden aangepakt om regelgevende acties en duurzame oplossingen te sturen. Hieronder volgt een uitgebreide samenvatting van deze drie delen:

DEEL A | Kennisoverzicht

Hoofdstuk 1 is de inleiding tot micro- en nanoplastics. Microplastics (MP) worden gedefinieerd als plastic deeltjes kleiner dan 5 mm, terwijl nanoplastics (NP) typisch variëren tussen 1 nm en 1 µm. Microplastics worden verder gecategoriseerd in primaire microplastics, die als kleine deeltjes worden vervaardigd voor specifieke toepassingen (bijv. microkorrels in cosmetica), en secundaire microplastics, die ontstaan door de fragmentatie van grotere plastic producten door milieuprocessen zoals verwering. De meest geproduceerde plastic polymeren die bijdragen aan microplasticvervuiling zijn onder andere polypropyleen (PP), polyvinylchloride (PVC) en verschillende vormen van polyethyleen (PE), zoals LDPE (lage dichtheid PE), LLDPE (lineaire LDPE) en HDPE (hoge dichtheid PE). Naast de basispolymeren bevatten kunststoffen vaak een reeks additieven—opzettelijk toegevoegde chemische verbindingen die zijn ontworpen om de prestaties te verbeteren. Veel van deze additieven worden beschouwd als zorgwekkende chemicaliën vanwege hun persistentie, bioaccumulatiepotentieel, mobiliteit in het milieu en toxiciteit (PBMT). Microplastics zijn zeer divers in structuur en gedrag. Deze deeltjes variëren sterk in vorm—waaronder vezels, fragmenten, korrels, sferen en pellets—en in samenstelling, omdat ze kunnen bestaan uit meerdere polymeren, additieven en maten. Deze complexe en variabele aard maakt microplastics een veelzijdig milieuprobleem, dat zorgvuldige studie van hun kenmerken en bronnen vereist.

In **Hoofdstuk 2** wordt de alomtegenwoordigheid van microplastics benadrukt, aangezien ze worden aangetroffen in lucht (8%), water (12%) en bodem (80%). Recent onderzoek door RIVM (Nederland) identificeert de meest significante bronnen van microplasticvervuiling als: bandenslijtage deeltjes (12 500 ton), plastic korrels gebruikt in de productie (primaire microplastics; 9 300 ton), en plastic afval dat afbreekt tot secundaire microplastics (8 600 ton) (Quik et al., 2024). Andere, kleinere bronnen zijn vezels van textiel (3 100 ton), opzettelijk geproduceerde microplastics (2 000 ton), verven en coatings (900 ton) en landbouw (880 ton). Hoewel landbouw relatief gezien een kleinere bijdrage levert, is het van bijzonder belang voor ons voedselsysteem. Hier kunnen microplastics afkomstig zijn van compost, zuiveringsslib (niet toegelaten in Vlaanderen), gecontroleerd vrijkomende meststoffen (CRFs), mulchen en irrigatieactiviteiten. Eenmaal in de bodem kunnen microplastics zowel horizontaal als verticaal migreren, wat zorgen baart over hun mogelijke intrede in de voedselketen en bredere ecosystemen.

In **Hoofdstuk 3** werden gegevens van meerdere studies over microplasticconcentraties in bodem, grondwater en sedimenten verzameld en samengevat. De bodemdatabase omvat 85 studies gepubliceerd tussen 2018 en 2024, met in totaal 199 bodemstalen uit verschillende regio's, met een primaire focus op Azië (57%) en Europa (27%). De studies richten zich voornamelijk op landbouwbodems (56%), met kleinere representaties van bos-, grasland-, kas-, moeras- en stedelijke bodems. De resultaten tonen aanzienlijke variatie in MP-concentraties, met 51% van de studies die minder dan 1 000 MP-deeltjes per kg bodem rapporteren, 37% tussen 1 000 en 10 000 deeltjes per kg bodem, en 12% die concentraties boven 10 000 deeltjes per kg bodem rapporteren. In grondwater varieert de concentratie van microplastics, gebaseerd op 58 studies uit 25 landen, van 0 tot 6 832 deeltjes per liter. Grondwater, als een vitale bron voor menselijke consumptie en het milieu, vormt een aanzienlijke zorg voor microplasticvervuiling, omdat het drinkwaterbronnen wereldwijd kan beïnvloeden. Voor sedimenten werd de MP-SED-database gebruikt, met gegevens van studies tussen 2004 en 2023. Onderzoek naar microplastics in sedimenten is voornamelijk uitgevoerd in Azië (32%), gevolgd door Europa (31%) en Noord-Amerika (30%), met verschillende omgevingen die zijn bestudeerd, zoals meren, rivieren en estuaria. De resultaten tonen aanzienlijke variatie in concentraties, variërend van 0 tot 74 800 MP-deeltjes per kg droog sediment, met de hoogste concentraties in gebieden met industriële activiteit. Overstromingen bleken de MP-concentraties aanzienlijk te veranderen, wat aangeeft dat omgevingsfactoren, zoals rivierconnectiviteit en atmosferische depositie, de verspreiding van microplastics in sedimenten in verschillende ecosystemen beïnvloeden.

Dit hoofdstuk benadrukt het belang van voortdurende studies die microplastics monitoren om hun wijdverspreide aanwezigheid en impact beter te begrijpen. Hoewel huidig onderzoek zich voornamelijk heeft gericht op landbouwbodems, is het cruciaal om de inspanningen uit te breiden naar bodems onder andere landgebruik, zodat een meer uitgebreide begrip van microplasticvervuiling in diverse ecosystemen wordt verzekerd.

Hoofdstuk 4 beschrijft de verschillende technieken voor het detecteren van microplastics in bodemmonsters worden, met de focus op zowel conventionele als nieuw opkomende methoden. Conventionele technieken omvatten doorgaans een reeks stappen, waaronder voorbehandeling (drogen), zuivering, MP-extractie, filtratie, identificatie en kwantificatie, met veelgebruikte methoden zoals FTIR-spectroscopie, Raman-spectroscopie,

visuele inspectie en het gebruik van kleurstoffen voor analyse. Opkomende methoden worden ook besproken, met name op het gebied van zuivering, extractie en detectie, met een tabel die de voor- en nadelen van elke techniek samenvat.
Tot op heden is er echter geen consensus over de beste analysemethoden, en blijven uitdagingen bestaan, vooral bij het omgaan met specifieke matrices (bijv. die rijk aan organisch materiaal) of rubberdeeltjes, die de analyse kunnen bemoeilijken. Binnen dit hoofdstuk ligt de focus op de kwaliteitsborging en kwaliteitscontrole (QA/QC), aangezien de kwantificatie van MP’s in milieu-monsters een uitdagende taak blijft, met problemen van contaminatie, overschatting en onderschatting die de resultaten beïnvloeden. Deze uitdagingen zijn vooral uitgesproken tijdens de extractiefase, waar juiste validatiestudies en het gebruik van blanco's essentieel zijn om betrouwbare gegevens te garanderen. Het hoofdstuk biedt een tabel die de essentiële QA/QC-procedures samenvat die helpen deze problemen aan te pakken en de nauwkeurigheid van de resultaten te verbeteren.

In **Hoofdstuk 5** verkennen we de effecten van MNP's op bodem, planten en menselijke gezondheid en het concept van veilige concentratielimieten van MP’s in verschillende omgevingen. MNP's kunnen belangrijke bodemkenmerken zoals porositeit, bulkdichtheid en waterbeschikbaarheid veranderen. De aanwezigheid van MNP's kan de bodemporositeit en bulkdichtheid verminderen, wat de bodemstructuur en waterretentie beïnvloedt, wat op zijn beurt kan leiden tot verhoogde verdamping en verminderde doorlaatbaarheid. De effecten van MNP's op bodemorganismen zijn even zorgwekkend, aangezien regenwormen, nematoden en geleedpotigen MNP-deeltjes kunnen opnemen, wat leidt tot verminderde groei, overleving en voortplanting. Microbiële gemeenschappen veranderen bij de aanwezigheid van MP’s, en daarnaast kunnen MP’s ook nieuwe niches creëren waarop microbiële biofilms kunnen vormen. Alles bij elkaar genomen worden effecten op bodemfuncties verwacht. In planten kunnen MNP's zowel directe als indirecte effecten hebben. Direct kunnen MNP's worden opgenomen door plantenwortels en bladeren, wat stress veroorzaakt en de groei belemmert. Kleinere deeltjes, in het bijzonder, hebben meer kans om plantweefsels binnen te dringen en aanzienlijke schade te veroorzaken. Indirect kunnen MNP's het microbiële ecosysteem van de bodem verstoren, wat de gunstige relaties tussen planten en micro-organismen, zoals stikstofbindende bacteriën en mycorrhiza-schimmels, kan verstoren. Deze interferentie kan de opname van voedingsstoffen verminderen en de plantengroei en productiviteit beïnvloeden.

De rol van MNP's in verontreinigingstransport wordt besproken in **Hoofdstuk 6.** MNP's kunnen verontreinigende stoffen adsorberen en fungeren als vectoren, waardoor ze door ecosystemen worden getransporteerd, waardoor immobiele verontreinigende stoffen mobiel worden en diepere bodemlagen of grondwater kunnen bereiken. De mogelijkheid van MP om verontreinigingstransport te verbeteren, wordt echter beperkt door factoren zoals desorptiesnelheden, bodemkenmerken en waterstroomomstandigheden.

Hoofdstuk 7 belicht het gedrag van MP’s in bodems en sediment, waarbij wordt benadrukt hoe bodemstructuur hun verspreiding en dynamiek beïnvloedt. MP’s kunnen worden opgenomen in bodemaggregaten, waarbij grotere macro-aggregaten meer deeltjes bevatten, terwijl bodemtextuur, MP-grootte en -vorm hun mobiliteit en migratie door het bodemprofiel beïnvloeden. Migratie wordt voornamelijk gedreven door bioturbatie van bodemorganismen zoals regenwormen en gewaswortels, met factoren zoals bodemporositeit en nat-droog cycli die hun transport beïnvloeden, mogelijk tot grondwater, hoewel dit niet overtuigend is bewezen in veldstudies.

In **Hoofdstuk 8** worden bestaande en opkomende regelgeving met betrekking tot MP-vervuiling in milieucompartimenten buiten de bodem, zoals mariene en zoetwatersystemen, besproken, samen met de huidige regelgevende hiaten in bodem, grondwater en sediment. Terwijl internationale inspanningen zich meer hebben gericht op aquatische omgevingen, blijven terrestrische compartimenten ondergereguleerd. De EU loopt voorop met het Zero Pollution Action Plan en recente verboden op opzettelijk toegevoegde MPs, maar de meeste landen behandelen MP’s alleen indirect, bijvoorbeeld door verboden op microbeads in cosmetica. Monitoring- en regelgevingskaders voor MP’s in bodem bevinden zich nog in een vroeg stadium, met beperkte nationale initiatieven zoals het Franse MICROSOF-project. Huidige regelgeving is gefragmenteerd en mist vaak een levenscyclusbenadering. Een wereldwijd verdrag over plasticvervuiling is in onderhandeling, met als doel de volledige levenscyclus van plastic aan te pakken en bindende doelen vast te stellen. Het hoofdstuk benadrukt de dringende noodzaak van gecoördineerde wereldwijde actie, sterkere regelgeving en geharmoniseerde monitoringnormen om MP-vervuiling effectief aan te pakken in alle milieucompartimenten.

In **Hoofdstuk 9** worden verschillende bodemsaneringstechnieken voor MP in bodems besproken, met de focus op hun mechanismen, toepasbaarheid en beperkingen. Er bestaan verschillende technieken om MP’s in bodems te saneren, hoewel geen enkele volledige verwijdering biedt, en hun effectiviteit varieert afhankelijk van MP-type, concentratie, bodemomstandigheden en gebruikte methode. Pyrolyse en fotokatalytische afbraak zijn ex-situ methoden voor toplaag, waarbij pyrolyse hoge MP-reductie biedt maar de bodem beschadigt en veel energie vereist, terwijl fotokatalyse minder invasief is maar beperkt in diepte. Magnetische extractie is een zachtere ex-situ optie met vergelijkbare efficiëntie. In-situ benaderingen omvatten fytoremediatie, waarbij specifieke planten worden gebruikt om MP’s te extraheren of te immobiliseren, maar lange cycli nodig hebben, en microbiële afbraak, die kosteneffectief en milieuvriendelijk is maar vertraagd door microbiële beperkingen en onzekere uitkomsten. Elke methode heeft context specifieke voordelen, zonder dat er één oplossing is die voor alle omstandigheden geschikt is.

DEEL B | Microplasticproblemen en -risico's

Het begrijpen van de risico's van MP’s voor ecosystemen en de menselijke gezondheid vereist meer dan alleen het meten van hoeveelheden. Belangrijke kenmerken zoals grootte, vorm, polymeertype, chemische additieven en zelfs de aanwezigheid van biofilms moeten ook worden overwogen. Experts benadrukken de noodzaak van een nieuw kader dat de complexe aard van microplastics volledig weerspiegelt. Om toekomstig onderzoek te verbeteren, moeten studies milieurelevante concentraties gebruiken en een mix van microplastictypes en -vormen omvatten. Het is ook belangrijk om ecotoxicologische tests uit te breiden naar verschillende bodemtypes en plantensoorten om beter de reële omstandigheden te weerspiegelen.

DEEL C | Toekomstige beleidsontwikkeling en belangrijke onderzoeken

Microplastics in bodems en grondwater zijn een opkomende bezorgdheid, maar onderzoek op dit gebied bevindt zich nog in een vroeg stadium. Het eerste hoofdstuk in DEEL C belicht de belangrijkste **kennisleemtes** die moeten worden aangepakt om effectieve monitoring- en mitigatiestrategieën te ontwikkelen. Belangrijke onbekenden zijn onder andere de werkelijke bronnen van microplastics—vooral van alledaagse consumptiegoederen en biogebaseerde plastics—hun concentraties in het milieu en hoe ze zich gedragen zodra ze in bodems of sedimenten terechtkomen. Huidige methoden hebben moeite om kleinere deeltjes te detecteren, waardoor het waarschijnlijk is dat MP-vervuiling aanzienlijk wordt ondergerapporteerd.

Er is ook beperkte kennis over hoe verschillende types, vormen en additieven in plastics ecosystemen en de menselijke gezondheid beïnvloeden. De meeste studies testen alleen individuele plastics onder vereenvoudigde omstandigheden, wat geen reële blootstelling aan complexe plasticmengsels weerspiegelt. Bovendien blijven de milieu- en gezondheidsimpact van duizenden plastic additieven grotendeels onbestudeerd en ongereguleerd.

In het tweede hoofdstuk van DEEL C worden **beleidsaanbevelingen** verkend. Een belangrijke prioriteit voor effectieve regelgeving is het verkrijgen van nauwkeurige inzichten in de huidige microplasticvervuilingsniveaus in bodems, water, lucht en voedsel. Deze gegevens zijn essentieel voor het beoordelen van potentiële risico's voor de menselijke gezondheid, plantontwikkeling en het milieu.

Ondanks de groeiende aandacht blijven er grote uitdagingen bestaan in hoe microplastics worden gemeten. Huidige technieken missen vaak kleinere deeltjes en variëren aanzienlijk tussen laboratoria. Om dit aan te pakken, moeten wetenschappers en beleidsmakers samenwerken om gestandaardiseerde methoden te ontwikkelen, regionale monitoringcampagnes te lanceren en voort te bouwen op bestaande Europese initiatieven zoals MiCoS, Papillons en MINAGRIS.

België bevindt zich in een sterke positie om het voortouw te nemen, gebruikmakend van bestaande netwerken zoals Cmon en LUCAS om nationale monitoring te starten. Idealiter zouden bodems elke 3 tot 5 jaar worden beoordeeld, met een focus op toplaaglagen en strikte bemonsteringsprotocollen om contaminatie te voorkomen.

De manier waarop microplasticvervuiling wordt gerapporteerd—op basis van deeltjesaantal of gewicht—hangt af van de gekozen methode en het onderzoeksdoel. Het harmoniseren van deze benaderingen zal essentieel zijn voor het produceren van vergelijkbare gegevens en het informeren van toekomstig plasticbeleid op zowel nationaal als Europees niveau.

KEY WORDS AND CONCEPTS

Table 1 presents the definitions utilized in this report. It is important to note that certain definitions (e.g. size classes) are subject to debate in the literature, as different sources may apply varying interpretations. Nevertheless, for the sake of clarity and consistency, this document adheres to the definitions outlined below.

Table 1 | Definitions related to microplastic research

	Definition
Macroplastics	Solid polymer particles larger than 2.5 cm in size, although sometimes the category of mesoplastics is also included in macroplastics.
Mesoplastics	Solid polymer particles ranging between 5 mm and 2.5 cm in size.
Microplastics	Solid polymer particles with an upper size limit of 5 mm.
Nanoplastics	Solid polymer particles smaller than 1 µm.
Plastic additives	Intentionally added chemical compounds incorporated into polymers during the manufacturing of plastics to enhance their properties and performance
Synthetic polymer	A human-made macromolecule composed of repeating structural units (monomers) chemically bonded through polymerization processes. These polymers are typically derived from petrochemical sources.
Primary microplastics	Microplastics intentionally produced for direct use or as raw materials, including: - Pre-production materials e.g. pellets, flakes and powders - Used directly as small pieces e.g. glitter, confetti Intentionally added to another product e.g. microbeads in cosmetics, pigments in paint
Secondary microplastics	Microplastics formed from the breakdown of larger plastic items, resulting from three sources: - Generated by wear and tear of products during use e.g. tyres, textiles, cleaning painted surfaces - Generated during waste management e.g. during recycling Generated by break down of larger items in the environment.
Elastomers	Polymers with viscoelastic properties, which are often called rubber.
Thermoplastics	Polymers that become mouldable when heated and solidify upon cooling, allowing for repeated reshaping.
Thermosets	Polymers that irreversibly harden after being heated and molded once, and cannot be remolded.
Oxo-plastics	Conventional plastics that are primarily polyethylene-based containing additives to accelerate oxidative degradation when exposed to light and/or heat.
Bioplastics	Is an umbrella term that encompasses a wide range of plastic materials made from renewable biological sources, including both biodegradable and non-biodegradable plastics
Biodegradable plastics	Plastics that can break down break down under certain environmental conditions, depending on the material and application.

Compostable plastics	Are a type of biodegradable plastic that decomposes under specific conditions, such as in industrial composting facilities. These plastics can be bio- or fossil fuel based

ABBREVIATIONS

Abbreviation	Full Term
µFTIR	Micro Fourier Transform Infrared
BPA	Bisphenol A
CFS	Coagulation-Flocculation-Sedimentation
CI	Confidence Intervals
CRF	Controlled Release Fertilizer
DOM	Dissolved Organic Matter
ECCC	Environment and Climate Change Canada
FPA	Focal Plane Array
FTIR	Fourier Transform Infrared
GC-MS	Gas Chromatography - Mass Spectrometry
HC5	Hazardous Concentration for 5% of Species
HDPE	High-Density Polyethylene
INC	Intergovernmental Negotiating Committee
IQR	Interquartile Range
IR-PHI	Full Infrared Photothermal Heterodyne Imaging
LAF	Laminar Airflow
LDPE	Low-Density Polyethylene
LLDPE	Linear Low-Density Polyethylene
LSPR	Localized Surface Plasmon Resonances
MEA	Multilateral Environmental Agreements
MNP	Micro- and Nanoplastic
MNS	Magnetic Nanoparticles
MP	Microplastic
MQ	Milli-Q (Ultra-pure Water System)
MSFD	Marine Strategy Framework Directive
NOAA	National Oceanic and Atmospheric Administration
NP	Nanoparticles
NPK	Nitrogen, Phosphorus, and Potassium
PA	Polyamide (Nylon)
PBAT	Polybutyleen Adipaat Tereftalaat
PBMT	Persistent, Bioaccumulative, Mobile, and/or Toxic
PE	Polyethylene
PES	Polyethersulfone
PET	Polyethylene Terephthalate
PEtG	Polyethylene Terephthalate Glycol-modified
PLA	Polylactic Acid
PMMA	Polymethyl Methacrylate (Acrylic)
POM	Particulate Organic Matter

PP	Polypropylene
PS	Polystyrene
PUR	Polyurethane
PVC	Polyvinyl Chloride
py-GC-MS	Pyrolysis - Gas Chromatography - Mass Spectrometry
QA/QC	Quality Assurance / Quality Control
SEM	Scanning Electron Microscopy
SERS	Surface-Enhanced Raman Spectroscopy
SSD	Species Sensitivity Distributions
TED-GC-MS	Thermal Extraction Desorption - Gas Chromatography - Mass Spectrometry
TRL	Technology Readiness Levels
UNEP	United Nations Environment Programme
UV	Ultraviolet
wt%	Weight Percent

PART A | KNOWLEDGE OVERVIEW

Research on plastic pollution began in 1974, initially focusing on aquatic ecosystems due to the high visibility of macroplastic pollution (> 2.5 mm). The study of microplastics (MPs, < 5 mm) in soil started much later. Following a 2012 paper by Rillig et al. suggesting a potential significant problem with (micro)plastics in soils, it took another five years for actual research on microplastics in soils to begin. Only in the last five years this research field has experienced rapid growth.

Within this report input is provided by University of Ghent, Department of Biochemistry and Microbiology, research group Micro2Soil embedded in LM-UGhent and Department of Environment, research group SoFer. External input of six experts on human and/or ecotoxicological effects on soil and plant health were collected via online interviews (Table 2).
The report is composed of three parts. PART A presents a general overview of the current knowledge of microplastics in soil and sediments. PART B assesses the extent of the microplastic problem and associated risks regarding eco- and human toxicological risks. PART C suggests future policy studies and policy-supporting research.

This report contains the opinion of the authors and not necessarily that of OVAM.

Table 2 | List of experts

Name expert	Institute	Country
Prof dr. Jana Asselman	Ghent University	Belgium
Prof dr. Barbro Melgert	Rijksuniversiteit Groningen	The Netherlands
Prof. dr. Willie Puijenburg	National Institute for public health and the environment (RIVM)	The Netherlands
Prof. dr. Bart Koelmans	Wageningen University and Research (WUR)	The Netherlands
Prof. dr. Ronny Blust	University Antwerp	Belgium
Prof. dr. Kees Van Gestel	Vrije Universiteit Amsterdam	The Netherlands

1 INTRODUCTION

Plastic pollution is a growing environmental concern, not only in the marine environment but also in soils and sediments, where its impact is understudied, but equally critical. Over time, large plastic debris breaks down into microplastics (MPs) through processes like weathering and (a)biotic degradation. These MPs accumulate in terrestrial environments, where they can persist for decades. Understanding sources, pathways, and impacts of microplastics in soils and sediments is crucial for mitigating this form of pollution.

1.1 WHAT ARE MICRO- AND NANOPLASTICS?

MPs are defined as solid polymer particles with an upper size limit of 5 mm, composed of polymers along with functional additives and other intentionally and unintentionally added chemicals (Thompson et al., 2024) (Figure 1). The term 'microplastics' was introduced by Thompson et al. (2004), although a consensus about a common size definition was lacking (Duis and Coors, 2016, Frias and Nash, 2019, Araujo et al., 2018, Horton et al., 2017, Van Cauwenberghe and Janssen, 2014, Courtney et al., 2008). The upper size limit of 5 mm was established by the NOAA in 2008 based on considerations of ingestion by organisms (Hartmann et al., 2019). While the upper limit is well-defined, the lower size limit is often set at 1 µm, however this remains constrained by methodological limitations in detecting and measuring smaller particles (Thompson et al., 2024). The smallest size class is named nanoplastics (NPs), which is defined by most authors as particles ranging between 1 nm and 1 µm (Gigault et al., 2018, Frias and Nash, 2019).

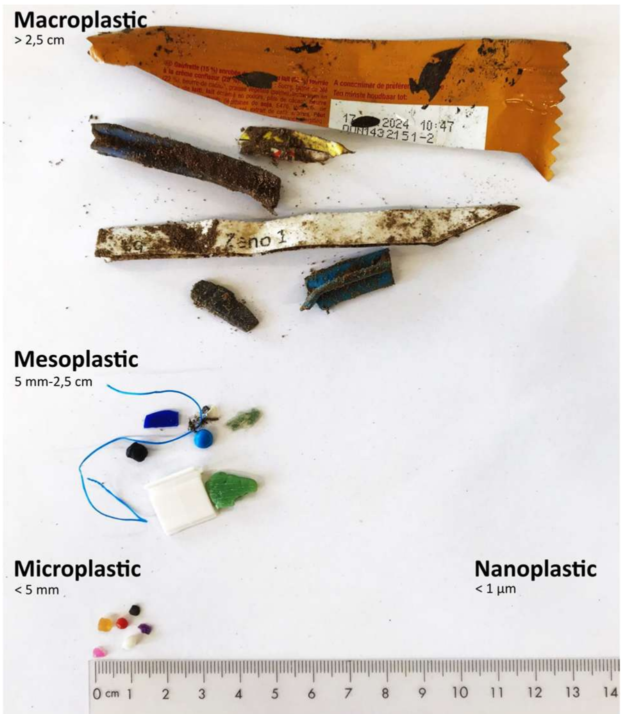

Figure 1 | Classification of plastic based on their size.

Subcategories of MPs have been described based on their origin, with the terms **primary and secondary MPs** being used. Thompson et al. (2024) proposes a universal framework for defining these subcategories, which we advocate as a consensus standard.

Primary MP are polymer particles that are manufactured as MPs (< 5 mm), which are either pre-production materials (pellets, flakes and powders), used directly as small pieces (glitter, confetti) or are intentionally added to another product (microbeads in cosmetics, toothpaste or industrial abrasives) (Duis and Coors, 2016, Kershaw and Rochman, 2016, Thompson et al., 2024). They are generally characterized by a homogeneous surface (Quik et al., 2022). **Secondary MP** result from the fragmentation and breakdown of larger plastic fragments (> 5 mm) in the environment (Cole et al., 2011, Rillig, 2012, Thompson et al., 2024, EFSA, 2021). The wear or use of products (textiles, tyres), loss of mechanical integrity (Kershaw & Rochman, 2015) or the influence of UV light (Andrady, 2011) in the environment in addition to fragmentation during waste management (recycling) are the main causes of plastic fragmentation (Thompson et al., 2024). Secondary MP typically possess a rougher surface and have a heterogeneous composition, which also influences their behavior in the environment (Quik et al., 2022). Most of the MP (70-90%) that end up in the environment are secondary MP (Verschoor and Van de Valk, 2018).

Micro- and nanoplastics (MNPs) exhibit a diverse range of shapes such as fibers, fragments, beads, spheres, pellets or granules, films, and foams (Hartmann et al., 2019) (Figure 2). Fragments are forms with an irregular shape. Beads, spheres and pellets are round particles which are often used as synonyms. However, both the terms beads and pellets refer to their origin (microbeads for cosmetics and preproduction pellets, respectively), which cannot always be ascertained. Films are planar objects, i.e. with one dimension being much smaller than the other two. Foams are a type of plastics with an expanded cellular structure. Fibers are threadlike plastics, i.e. with a pronounced elongation (i.e. a very large length-to-diameter ratio).

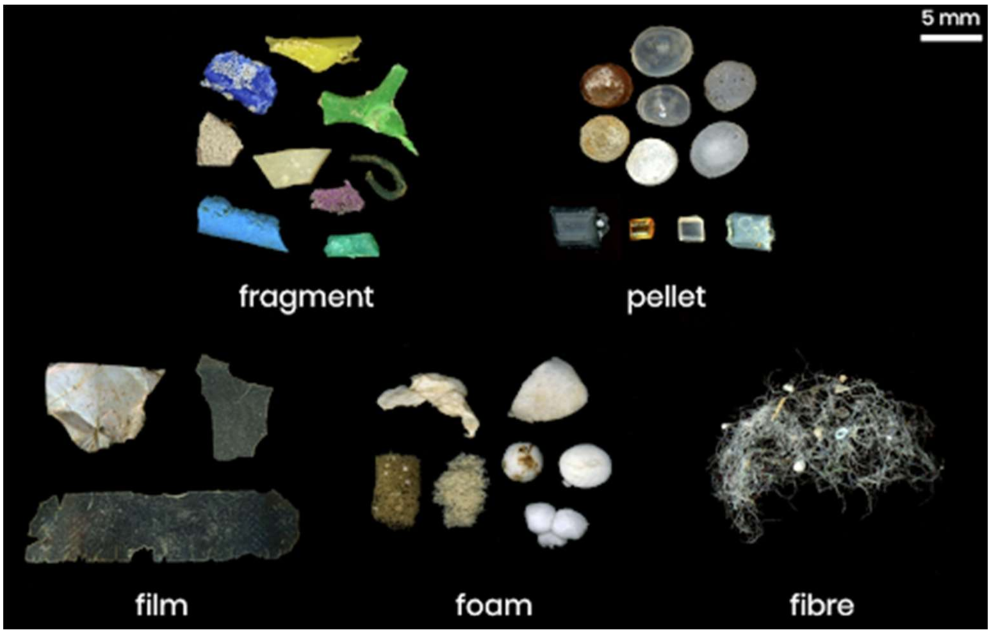

Figure 2 | Shapes of microplastics. Ranging from fragment (irregular shape) to pellets (round particles), films (planer objects with one dimension smaller than the other two), foams (expanded cellular structure) and fibers (threadlike). Source: Kunz (2022).

1.2 WHICH TYPES OF PLASTICS ARE RELEVANT?

The global production of plastics has reached 413.8 Mt in 2023 (Figure 3), which is an increase of 3.4% compared to 2021 (Plastics Europe, 2024). Plastics are categorized based on their chemical composition, which also influences their behavior and potential impacts on soil ecosystems. The most produced plastics are polypropylene (PP), polyvinyl chloride (PVC) and polyethylene (PE), which is further divided based on its density, namely low density (LDPE), linear low-density (LLDPE), and high-density (HDPE) (Figure 3).

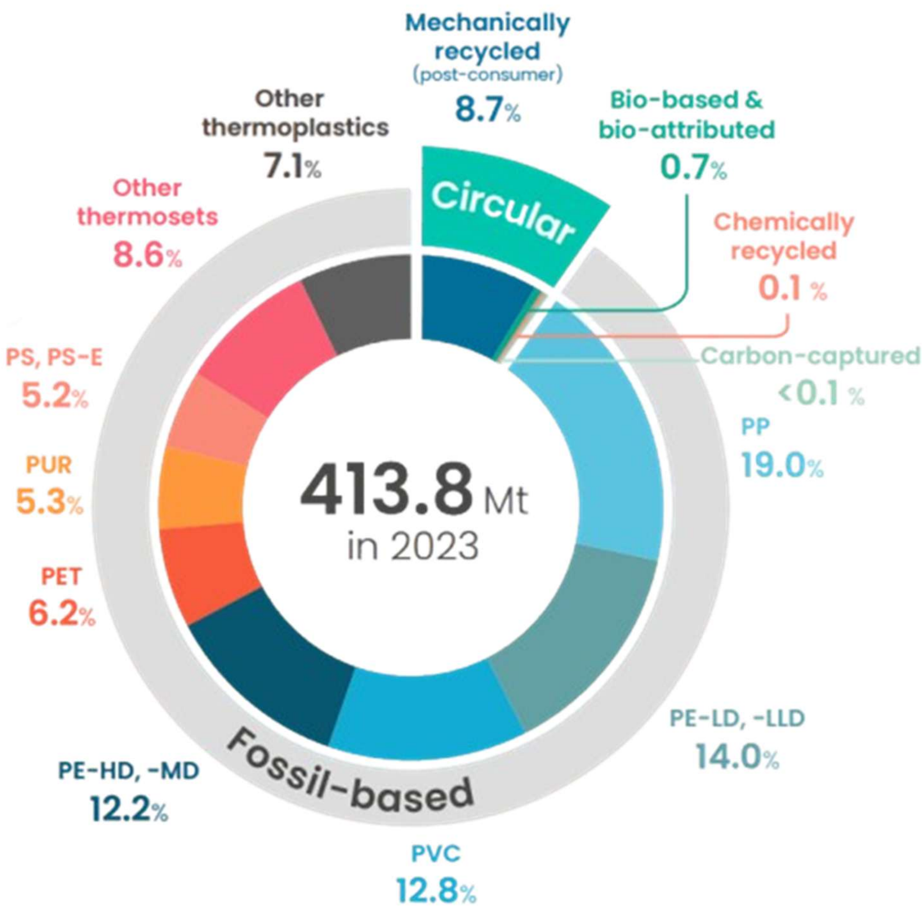

Figure 3 | Production rates of plastics in 2023. *PS* (Polystyrene), *PS-E* (Expanded Polystyrene), *PUR* (Polyurethane), *PET* (Polyethylene Terephthalate), *PE-HD/MD* (High- and Medium-Density Polyethylene), *PVC* (Polyvinyl Chloride), *PE-LD/LLD* (Low- and Linear Low-Density Polyethylene), *PP* (Polypropylene). *Thermoplastics* are plastics that can be melted and reshaped multiple times, while *thermosets* undergo irreversible curing. *Mechanically recycled* (post-consumer) plastics are reprocessed from used plastic waste. *Bio-based plastics* are derived from renewable biological sources, whereas *bio-attributed plastics* combine fossil and renewable feedstocks with certified attribution. *Chemically recycled plastics* are broken down into monomers or other raw materials for reuse. *Carbon-captured plastics* incorporate carbon dioxide or other captured emissions into the production process. Source: Plastics Europe (2024).

Table 3 provides an overview of the most frequently produced plastics found in soils, as reported by (Wahyudi and Cordova, 2016). These include polyolefins such as PE, Polyethylene terephthalate (PET), Polystyrene (PS), and PVC. Additionally, Polyamide (PA), Polymethyl Methacrylate (PMMA), and synthetic rubbers play an important part in environmental pollution. Polyurethane (PUR) constitutes 8% of all of the produced polymers on a global scale (PlasticsEurope, 2019).

Table 3 | Overview of the most frequently produced plastics found back in soil.

Polymer	Abbreviation	Explanation
Polyethylene	PE	Used in plastic bags, bottles, and packaging materials. It's one of the most commonly encountered MPs in marine environments.
Polypropylene	PP	Common in food containers, straws, and bottle caps. It is another common form of microplastic found in the environment.
Polystyrene	PS	Found in items like foam cups, plates, and packaging materials. Expanded polystyrene (EPS) foam is particularly problematic.
Polyethylene terephthalate (PET)	PET	Common in plastic bottles, textiles, and packaging. PET MPs are often found in marine environments.
Polyamide	PA	Often used in textiles (like nylon) and in industrial applications. Nylon fibers in particular are shed in large quantities from synthetic clothing.
Polyvinyl chloride	PVC	Construction, packaging, clothing and upholstery, medical equipment, electric insulation.
Polyurethane	PUR	Automobiles, coatings, textiles and medical applications (implants, medical devices and drug-controlled release carriers).

Beyond conventional fossil fuel-based plastics, the umbrella term “**bioplastics**” includes all plastics that can be biobased, fossil-based, biodegradable or non-biodegradable (Figure 4). Importantly, bioplastics are not necessarily biodegradable or compostable, and they may still contain fossil-derived components. **Biobased plastics** are derived, either partially or entirely, from renewable biological sources. Common feedstocks are starch, sugarcane or cellulose (EuropeanBioplastics, 2024). **Biodegradable plastics** break down under certain environmental conditions, depending on the material and its intended application (Paço et al., 2018, Peng et al., 2022). In addition, **compostable plastics**, a subset of biodegradable ones, typically require high temperatures to decompose effectively. These conditions are only met in industrial composting facilities and are not guaranteed in home composting. Biodegradable and/or compostable plastics do not necessarily equal biobased as they can also be produced from fossil raw materials. While biodegradable plastics may offer some reduction in long-term pollution, their behavior in different environments, such as soils, remains a subject of ongoing research to fully understand their ecological impacts. Additionally, compostable plastics often disintegrate into MPs rather than fully degrading, potentially exacerbating the microplastic pollution problem.

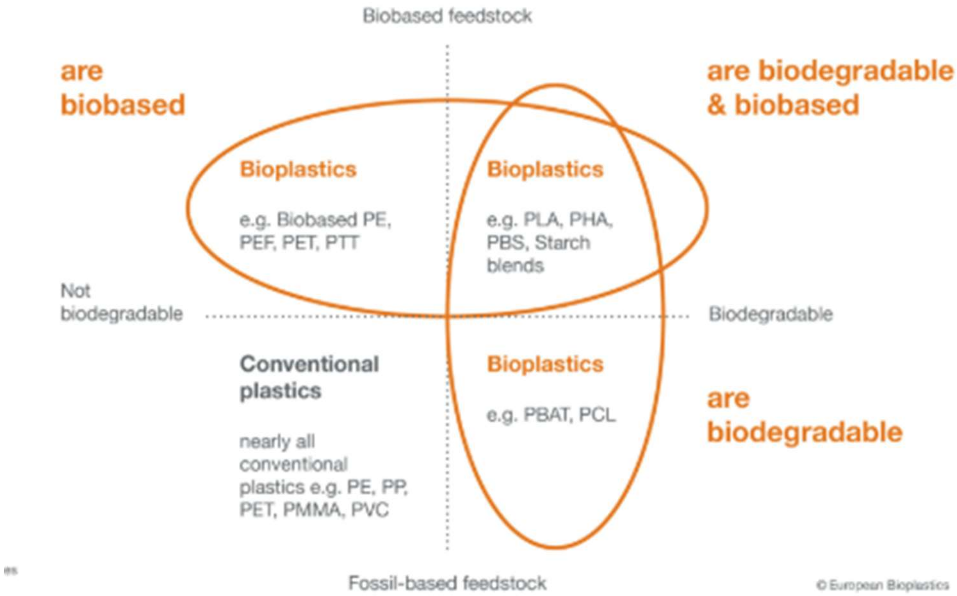

Figure 4 | Classification of plastics based on their feedstock (biobased or fossil-based) and biodegradability. The diagram highlights that the term bioplastics is used for multiple categories, including biobased but non-biodegradable plastics, biodegradable and biobased plastics, as well as fossil-based biodegradable plastics. Source EuropeanBioplastics (2024).

The relevance and prevalence of specific polymer types in the environment vary significantly depending on the environmental context (marine, freshwater, soil, other) and sector (e.g. agriculture). Rather than attempting to cover all polymers in detail, this section will focus on the most relevant plastic types in key sectors, such as agriculture, urban environments, and transportation infrastructure.

In agricultural settings, LDPE and PVC are the most commonly found plastics in agricultural soil (Quik et al., 2024, Hofmann et al., 2023). LDPE is frequently used in agricultural mulching, agricultural pipes, greenhouse films, ..., while PVC is primarily associated with agricultural pipes. Use of PP and PS is much less common in this sector. When focusing on soil pollutants, a focus on PVC and LDPE should therefore be considered. In addition, biodegradable alternatives such as polylactic acid (PLA) and PBAT are increased in usage in the agricultural sector and should therefore be considered as well in terms of their effects on soil health, plant yield and microplastic pollution.

Along roadsides the most produced polymers, including PE, PP, PES, PS are found the most as waste (Monira et al., 2022, Su et al., 2020). Therefore, it could be considered that these polymers will also be present in highest concentrations as MPs. In urban soils, PP, PE, PVC, and rubber are identified the most (Leitao et al., 2023).

As the number of (bio)plastic polymers is high, and it is expected that (1) each of the polymers itself will affect the soil and plant on a different way, (2) each will contain different additives (see Chapter 1.3) and (3) each will contain a different microbial community, one should take into account the environmental context and production rate when focusing on the microplastic effects. Within this document and based on the above given documentation, we therefore focus primarily on highly produced and recovered MPs in the above described area (PE, PP, PVC, rubber) and newly rising bioplastics (PLA, PBAT).

1.3 WHICH ADDITIVES PLAY A ROLE IN THE MICROPLASTIC STORY?

Plastic additives are intentionally added chemical compounds incorporated into polymers during the manufacturing of plastics to enhance their properties and performance (**Table 4**), including plasticizers, polymer stabilizers, and flame retardants (Bridson et al., 2021, Zweifel, 2009). Other additives are often related to aid in the manufacturing process (e.g. anti-static agents or lubricants). The type of additives that is present in a specific product depends on the type of polymer, the intended application, and the desired properties.

Table 4 | Overview of additives, their function and examples. Sources: Zweifel (2009), Bridson et al. (2021).

Type	Function	Examples
Plasticizers	Increase flexibility and reduce rigidity of plastics.	Phthalates (e.g., di-2-ethylhexyl phthalate, DEHP), adipates, citrates.
Stabilizers	Protect the plastic from degradation due to UV radiation, heat, or oxygen exposure.	Benzophenones, Hindered Amine Light Stabilizers (HALS), Thermal stabilizers: Zinc or lead compounds, organic stabilizers like zinc stearate.
Flame retardants	Reduce the flammability of plastics.	Halogenated flame retardants (e.g., tetrabromobisphenol A). Phosphorus-based flame retardants (e.g., tris(2-chloroethyl) phosphate). Intumescent flame retardants (e.g., ammonium polyphosphate).
Colorants and pigments	Provide color to the plastic	Organic and inorganic colorants such as titanium dioxide (white) and carbon black.
Fillers	Increase the strength, rigidity, or cost-efficiency of plastic by filling volume.	Glass fibers, calcium carbonate, talc, mica.
Antioxidants	Protect the plastic from aging caused by oxygen or heat exposure.	Butylhydroxytoluene (BHT), butylhydroxyanisole (BHA), sterically hindered phenols, 6PPD.
Antistatic agents	Prevent static electricity buildup in plastics.	Quaternary ammonium compounds.
Nucleating agents	Promote better crystallization, improving the mechanical properties of plastics (e.g., in PET or HDPE).	Organic acid salts such as benzoic acid or adipic acid
Melt flow improvers	Affect the melting behavior of plastics to aid in processing	Glycol or polyethylene glycol compounds
Blowing agents	Help in the formation of foamed plastic structures.	Azodicarbonamide, phosphates
Surfactants	Improve the dispersion and adhesion of various components in plastic formulations.	Polysiloxanes, stearyl alcohol
Lubricants	Reduce friction and improve processability during the production of plastics.	Stearates, PEG (polyethylene glycol)

Polymerization inhibitors or retarders	Control the polymerization rate in certain plastic processes, such as in PVC or polyurethane production.	Hydroquinone, butylhydroxyanisole
Biocides	Prevent mold and bacterial growth in plastics.	Silver compounds, copper compounds, zinc pyrithione

While a wide array of chemicals is applied during the process, many of them are inadequately studied (Wagner et al., 2024). The PlastChem project (plastchem-project.org) recently published a report that provides an overview of 16 325 chemicals potentially used or present in plastic materials and products. Only 6% of them are currently internationally regulated. More than 4 200 of these plastic chemicals are of **concern** because they meet one or more criteria of being **persistent, bioaccumulative, mobile, and/or toxic (PBMT)**[1]. Of these, 1 300 are being intentionally added to plastics during the manufacturing process. Among these, 15 groups of plastic chemicals, comprising 795 substances, are identified as major concerns and would be subject to regulation in e.g. multilateral environmental agreements (MEA). This includes chemicals from the groups of aromatic amines, aralkyl aldehydes, and aromatic ethers, all of which are classified as hazardous (**Table 5**).

Table 5 | Plastic chemicals of concern. Source: Wagner et al. (2024).

Aromatic amines	Bisphenols	Parabens
Aralkyl aldehydes	Phthalates	Azodyes
Alkylphenols	Benzothiazoles	Aceto/benzophenones
Salicylate esters	Benzotriazoles	Chlorinated paraffins
Aromatic ethers	Organometallics	Per- and polyfluoroalkyl substances (PFAS)

These chemicals can be released back into the environment during various stages of plastic product's lifecycle, including production, use, disposal and the weathering of plastics in soils. When addressing the issue of plastic pollution, it is essential to also consider plastic additives and, by extension, non-polymer plastic chemicals. Failing to account for the chemical aspects of plastics will hinder efforts to prevent and mitigate their negative environmental impacts and soil health.

The additives that are most likely to end up in the environment are those that are persistent, toxic, or bioaccumulative, such as plasticizers (phthalates), flame retardants (PBDEs), certain biocides (e.g., silver nanoparticles), and antioxidants (Zhang et al., 2024, Maddela et al., 2023). From the PlastChem report, rubber, PUR, ABS, PC, and PVC are most likely to contain chemicals of concern, additionally for PET, PE and PP more than 100 chemicals of concern are also released (Wagner et al., 2024) (Figure 5).

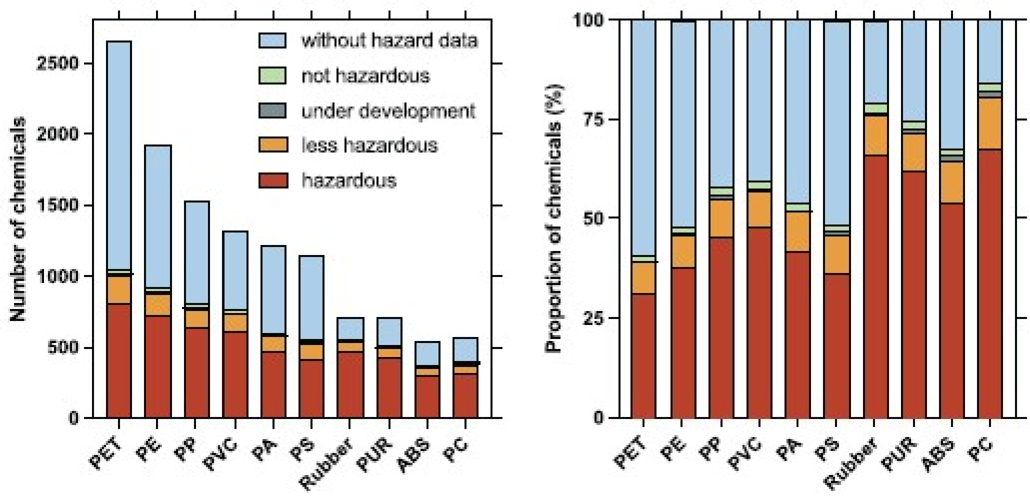

Figure 5 | Hazard information for chemicals used or detected in plastics according to polymer type. Left: number of chemicals, Right: proportion of chemicals normalized to the number of chemicals used or detected in each group. Source: Wagner et al. (2024).

2 THE EXTENT OF MICROPLASTIC POLLUTION

2.1 WHAT IS THE EXTENT OF MICRO- AND NANOPLASTICS CONTAMINATION IN SOIL AND SEDIMENTS?

MPs are ubiquitous pollutants that have invaded virtually every environment on Earth (Huang et al., 2021, Saini and Sharma, 2022). Their widespread distribution is evident across all compartments of the environment (air (8%), water (12%), and soil (80%)) (Quik et al., 2024). They have been found even in the most remote terrestrial environments that are supposed not to be under anthropogenic disturbances, including remote deserts, peaks of mountain ranges, pristine forests, the Artic and Antarctic regions. In this way, MP are amongst the most prominent indicators of global change. Their ubiquity can be attributed to their small size, allowing them to travel very large distances in the atmosphere and be deposited thousands of kilometers from their source. It is safe to say that MP would be detected in every environment in which they are looked for, and that the only reason for negative results (not finding MP) is the limitation in current detection methods of MP, which in soils are typically limited to several tens of micrometers. Since all water bodies on Earth are contaminated by MPs to varying degrees, it follows that the sediments within these water bodies also must be contaminated to some extent.

2.2 WHAT ARE THE MAIN SOURCES (HISTORICAL AND CURRENT)?

A comprehensive study by the RIVM (Rijksinstituut voor Volksgezondheid en Milieu) identifies the main sources of MPs entering the environment (Quik et al., 2024). According to this report, the three primary sources of MPs are (i) tyre wear on road surfaces, (ii) plastic granules used by industry to produce plastic products (primary MP), and (iii) plastic waste, including macroplastic litter which fragments into secondary MP (Quik et al., 2024) (Figure 6). Smaller sources of MPs pollution include textiles from clothing, paints, certain pesticides, rubber granules used for artificial turf fields and plastics used in agriculture (Quik et al., 2024). In the following sections, these emissions sources will be discussed in more detail.

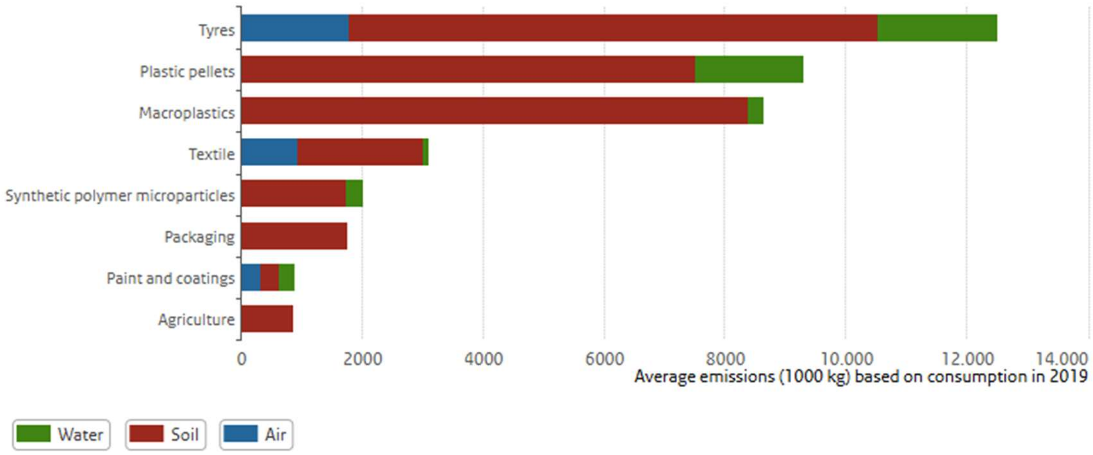

Figure 6 | Main sources of microplastics, and their contribution to emissions to air, soil and water. Sources RIVM (2024).

2.2.1 Tyre wear

The most important source of MP is believed to be from road tyre wear, so-called Tyre Wear Particles (TWP), reported to make up 56% of all MPs (Ateia et al., 2022, Sundt et al., 2014). In the Netherlands, two recent studies gave estimates of annual TWP emissions. Quik et al. (2024) estimated annual TWP emission ranging between 7 500 and 19 000 t. The lower end of this range is supported by Hoeke et al. (2024), who estimated emissions at 7 800 t/year in the Netherlands. However, the latter study excludes TWP emissions into the air, and sees them as only contributing marginally. On the other hand, Urbanus et al. (2022) reported much lower annual emissions of on average 2 600 t/year.

Tyre wear particles enter the environment through multiple pathways. An estimated 5-10% is emitted into the air (Gerben, 2022, Hoeke et al., 2024, Verschoor et al., 2016), while the remainder is distributed as follows: a portion is removed by street cleaning, which varies by road type (80-90% for ZOAB (highly porous hot-rolled asphalt) roads and 1-2% for other roads) (Sieber et al., 2020), some particles are resuspended into roadside soil due to traffic (Verschoor et al., 2016), and others are transported by rain runoff (Hoeke et al., 2024). Road runoff is known to transport pollutants such as suspended solids, heavy metals, and herbicides in a fast way towards the surface waters (Barbosa and Fernandes, 2021, Crabtree et al., 2006). Road runoff can also be considered an important pathway for MP transport towards soils and watercourses, since tyre wear is a capital source of MPs (Boucher and Friot, 2017). However, 36% of the TWP does not reach the environment, but is extracted by road cleaning (highways) or water treatment (urban areas) (Quik et al., 2024).

Tyre wear transported by rain runoff follows different pathways, ending up in (a) wastewater in combined sewer systems, (b) stormwater in separated sewer systems, (c) directly in surface water, or (d) roadside soil (Hoeke et al., 2024).

2.2.2 Plastic pellets

Primary MP constitute a major source of MP pollution into the soil and consists of two important categories, namely pre-production pellets and intentionally produced polymer microparticles. Dutch estimates for the release of pre-production pellets to the environment are slightly lower than those for TWP: between 6 900 to 12 000 t/year (Quik et al., 2024). These emissions result from losses that can occur at the production site, but also during overland transport, sea transport or the recycling process. Losses from the first two categories are the most significant. However, Verschoor and Van de Valk (2018) estimated pellet losses of about 1 000 t of these losses are released to the water, which is around 50% of the total of approximately 1 800 t that is released to the environment in this manner.

Specifically for Belgium, the Port of Antwerp as the leading polymer hub in Europe for the production, handling and transportation of pre-production plastic pellets is identified as a hotspot for plastic pellet pollution. This is a major source of plastic pellets entering the Scheldt estuaries, riverbanks and entering the North Sea. The major sources are PE and PP as these are the primary produced plastics in Antwerp.

Dutch emissions from intentionally produced polymer microparticles result from infill material (550-800 t) and offshore oil and gas industry (50-390 t). Only around 200 t (100-320 t) originate from daily consumer products like detergents and maintenance products, personal care products, food additives and medical applications (Quik et al., 2024). A very large contribution of intentionally produced polymer microparticles is related to agriculture and will be discussed under inputs from agricultural activities.

2.2.3 Plastic waste

About 8 600 t of macroplastics are emitted to the environment in The Netherlands, mostly originating from packaging, textiles and agricultural applications of plastics (Quik et al., 2024) and are the main source of plastic litter in the environment (Urbanus et al., 2022, Kawecki and Nowack, 2019). However, a recent report (Stegmann et al., 2024) estimated that about 14 000 t of packaging plastic alone was lost in the environment in the Netherlands for 2017, which might indicate that the numbers of plastic waste are being strongly underestimated. There are no data for Belgium about how much plastic waste is lost to the environment. In 2022, however the total amount of waste reached 63 Mt, of which construction and the industry were the main producers (Statbel, 2022). Households accounted for 7.7% of waste, while agricultural activities only accounted for 0.4% of the total waste production (Figure 7).

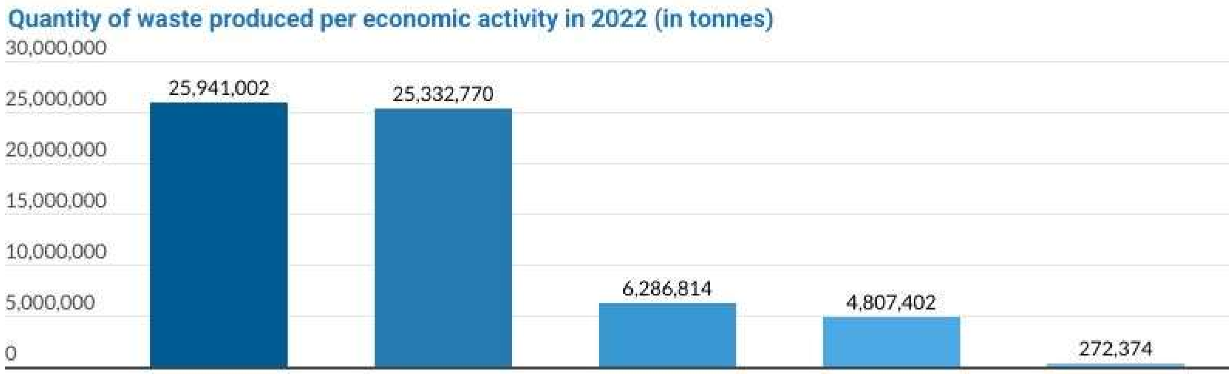

Figure 7 | Total number of plastic waste produced per sector. Construction and industry together are the main polluters, with over 80% of plastic waste generated. On the other hand, agriculture only resulted in 0.4% of plastic waste. Source: Statbel (2022).

2.2.4 Agricultural activities

Quik et al. (2024) estimate emissions from agriculture in the Netherlands as relatively low compared to the sources mentioned above, averaging 880 t annually. However, plastics are used and handled during agricultural production. Mulching films and greenhouses are used to improve crop yield and quality, while irrigation pipes ensure efficient crop watering. The primary plastics used are PE, PVC, and, to some extent, PS and PP (Piehl et al., 2018, Quik et al., 2024). This is also reflected in the pollution numbers as PE is the most found plastic in agricultural soils, next to PS and PP (Piehl et al., 2018). Agricultural activities such as ploughing can contribute to the fragmentation and dispersion of plastics. Additionally, the application of controlled-release fertilizers (CRFs), compost or sewage sludge can introduce (micro)plastics into agricultural soils.

CRFs are nutrient pills usually with non-degradable polymer coatings releasing nitrogen, phosphorus and potassium (NPK) into the soil during a given period, securing a constant supply of nutrients to plant roots (Goertz, 2000). Quik et al. (2024) estimate emissions from CRFs for The Netherlands to range between 710 - 1 210 t/year. The polymer of the coating remains in the soil (Gionfra, 2018, Kershaw and Rochman, 2016). The most used polymers for CRFs are polyacrylate, vinyl-alcohol, starch-based, polysulfone, chitosan, polyhydroxybutyrate, κ-carrageen, polystyrene, and polycaprolactone (Fitri et al., 2021). In recent years, CRFs have become available without plastic coating, for instance by using polyglyoxylate-polyester (PEtG) blends responsive to root-driven pH changes, promoting plant growth comparable to existing CRF (Heuchan et al., 2019). Bio-nanocomposites derived from renewable sources are also being investigated as potential coating materials (Vejan et al., 2021).

MPs are prevalent in various types of **compost**, with concentrations ranging from 7 to 1 315 particles/kg dry weight (Zafiu et al., 2023, Berset et al., 2024). The quantities and types of MPs that are present depend on the type of compost (green compost, organic waste, farm compost, composted manure, other). Commercial composts were found to contain up to 800 MPs/kg (Iswahyudi et al., 2024). MPs are found in different sizes (0.05-5 mm), shapes (fragments, fibers, films), and colors, with PE, PP, and PE being common polymer types (Iswahyudi et al., 2024, Gui et al., 2021). The composting process can generate and fragment MPs, with more frequent turning potentially increasing MP generation (Zafiu et al., 2023). MP concentrations vary between compost types, with those containing organic household waste showing higher levels (Berset et al., 2024). Rural domestic waste compost is a significant source of MPs in soils, with macroplastics potentially releasing 4-63 MP particles during composting (Gui et al., 2021). These findings highlight the need for careful management of compost production, through both the pre-screening of biowaste and post-processing of compost in order to reduce MP pollution in agriculture.

In many European countries, fertilization with **sewage sludge** is thought to be one of the most important contributors to plastic pollution in soil (Radford et al., 2023). Rolsky et al. (2020) report an average particle count in sludge from 12 countries of 12.8 ± 5.2 MP/ g sludge, with PE, PP and PA as the most commonly found polymers (Hassan et al., 2023). However, in Flanders, legal constraints prevent the use of sewage sludge as a soil amendment.

LDPE, HDPE, and LLDPE are the most common types of plastic used for **mulching** (Steinmetz et al., 2016). Some mulches are made from oxo-plastics, which are conventional primarily PE-based plastics with additives that accelerate oxidative degradation under light and heat. However, mulches that contain non-biodegradable plastics constitute an important source of microplastic pollution. since they are not collected after use (Thomas et al., 2012). Hu et al. (2021) investigated the distribution of MPs in plastic mulched soils in Xinjiang, China, and found on average 161.5 MP fragments/100 g in the 0-30 cm soil layer, mostly consisting of fibers, fragments, and particles. There is a clear decrease in depth, with only 11.20 ± 1.10 pieces/100 g present between 40-80 cm. However, this study does not give an overview of the relative contribution of the plastic mulch to the plastic pollution. Recent research in Spain has revealed that plastic mulch contributes significantly to plastic pollution in soil, with quantities up to 60 cm^2/kg and 0.2 g/kg. Assuming a mulch application of 0.9 ha mulch/ ha field per year, this represents 10-20% of the total mulch that is left as debris in the soil applied over 25 years (Beriot et al., 2023).

Irrigation activities can potentially cause microplastic pollution (Bläsing and Amelung, 2018), but concentrations remain unknown to this day. Often, irrigation water contains large amounts of MPs (Bläsing and Amelung, 2018, da Costa et al., 2016, Hurley and Nizzetto, 2018). Several sources of water can be used for irrigation, going from rainwater to groundwater and wastewater. In a recent study conducted in the MiCoS project, it was noted that at least for meso- and macroplastic pollution, the type of irrigation did not play a major role in differences in pollution (data unpublished). Nevertheless, it can be expected that for MPs especially irrigation with grey water can be seen as a main contributor. According to Mateo-Sagasta et al. (2013), 20 million hectares are irrigated with untreated wastewater worldwide. A recent study in Flanders showed that **even in wastewater treatment plants, a high amount of** MPs is present with PS (58%), PP (19%) and PET (12%) as main contributors. In addition, despite the high removal efficiency (97.5%), still 1.11 x 10^7 MPs, predominantly smaller particles (25-75 µm), end up in the nearby waterways on a daily base in the form of fibers and fragments, which eventually can be released in the sediment and soil (Vercauteren et al., 2023b). The differences in concentrations between treated and untreated wastewater are therefore considerable, ranging from 0 to 125 000 MP/ m^3 and 1 000 to 627 000 MP/ m^3 respectively (Carr et al., 2016, Talvitie and Heinonen, 2014).

2.3 HOW DO MICROPLASTICS END UP IN SOIL, GROUNDWATER, SEDIMENT, PLANTS AND THE FOOD CHAIN?

For soil ecosystems, tyre particles, plastic waste, plastic pellets, but also agricultural activities including the use of CRFs, compost, sewage sludge, mulching and irrigation are the main sources, as described above (Chapter 2.2).

Both for water, sediments and soils, part of the microplastic pollution originates from domestic products like household cleaning products, personal care and cosmetic products, paint, coatings or textile washing. Geyer et al. (2017) estimate that by 2050 up to 12 000 Mt of plastic waste will be deposited worldwide in landfills and in the natural environment. According to Browne et al. (2011) the presence of MPs in **municipal wastewater l**argely originates from clothing, the synthetic fibers (viscose, acrylic and nylon) of which are released during washing and end up in the wastewater. Cosmetics also add to the presence of primary MPs in the environment (Carr et al., 2016). Up to 56% of MPs reportedly originates from tyre wear (Ateia et al., 2022, Sundt et al., 2014).

MPs migrate horizontally and vertically in soil, potentially entering the **food chain** eventually. It has been shown previously that MPs can be taken up by the **plant** through root transport, in which it can accumulate in the leaves, the fruits and even the plant cells (Liu et al., 2022b, Li et al., 2019a). These MPs can then be further transported into the food chain as plants will serve as feed for animals (both in soil and marine environments), earthworms (degradation of plant leaves) and nematodes. In addition, MPs can accumulate in organisms through processes like filter feeding (in marine environments, e.g. oysters) and ingestion, with concentrations increasing at higher trophic levels (Saeedi, 2023). Despite the widespread use of plastics in food packaging, research on MPs in the food chain remains insufficient and requires further attention to protect public health (Cverenkárová et al., 2021).

3 CONCENTRATIONS OF MICROPLASTICS IN SOILS, SEDIMENTS AND GROUNDWATER

3.1 WHERE DO YOU EXPECT TO FIND THE HIGHEST MICROPLASTIC CONCENTRATIONS AND CAN THEY BE RELATED TO CERTAIN ACTIVITIES (HISTORICAL AND CURRENT) OR DESTINATION TYPES?

The presence of microplastics in the terrestrial ecosystems is linked to various anthropogenic activities, coupled to the main emission sources as outlined in Chapter 2. This pervasiveness is linked to both historical and current human activities. Since the 1960s, an industrial boom has led to a dramatic increase in plastic production, significantly contributing to microplastic pollution. Rapid urbanization and industrialization have further accelerated plastic waste generation and its widespread dispersal. These activities contribute significantly to the accumulation of MPs in these ecosystems. The proportion of MNP in the total amount of plastic waste is steadily rising and is estimated up to 13.2% in total weight by 2060 (Sharma et al., 2021). MPs have been detected in all environmental compartments from the most remote to the most urbanized and industrialized areas (Akdogan and Guven, 2019). The fate of plastic is concerning, as only a minimal amount is recycled, leaving large quantities to be both intentionally and unintentionally introduced into the terrestrial environment. Within MP research it is necessary to incorporate questions related to current and future plastic pollution. This data is however difficult to collect as land-use changes over time (Palazot et al., 2024). The underlying driver in all cases are human activities (Thompson et al., 2024).

Several hotspots for MPs (in soils) have been identified, though this list is not exhaustive:

- Urban areas and exploited areas
- Industrial areas
- Landfills
- Agricultural soils
- Sediments

Urban and exploited areas are key receptors of plastic pollution due to their high population densities and intensified human activities (de Souza Machado et al., 2019). Between 1950 and 2015 around 4 900 Mt of plastic garbage was amassed in **landfills** and natural environments worldwide (Geyer et al., 2017). Belgium is the number one worldwide in plastic waste recycling (> 35% of plastic recycled), and a small fraction (< 2%) diverted to landfills (Figure 8). Nevertheless, littering from urbanization and landfills remain significant, even within Europe, and it is expected to contribute to microplastic pollution. However, to date, no research has been done to quantify the amount of microplastic that reaches the soil through littering or illegal dumping (Yang et al., 2021b). Litter can also end up in the soil by wind blowing, street runoff and flooding. For instance, Büks and Kaupenjohann (2020) report that soil concentrations in both China and Europe are up to ten times higher in municipal areas than in rural ones with the highest values being recorded in industrial areas. Extreme values on industrial sites exceed common concentrations by 2 to 4 orders of magnitude (Büks and Kaupenjohann, 2020).

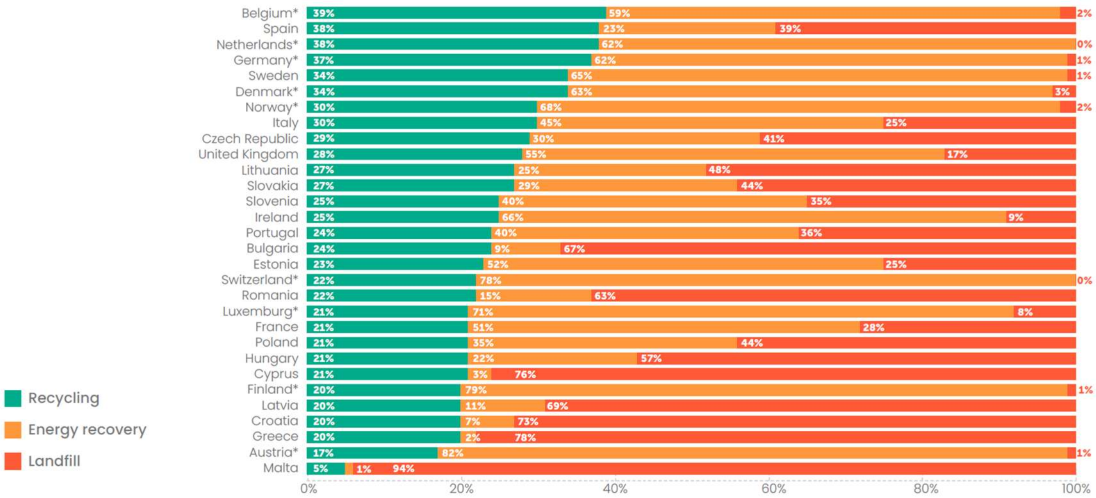

Figure 8 | Plastic waste treatment by country. (Source: PlasticsEurope (2024))

According to expectations, studies conducted in **natural ecosystems,** including forests, peatlands, and natural grasslands, often report very low concentrations of MPs compared to areas near urban activities, or fail to identify any MP. For example, a study in French soils found only in one of the four studied forests the presence of MPs (Palazot et al., 2024). MPs have also been identified in remote natural areas far from direct pollution sources, highlighting their ability to infiltrate diverse environments through mechanisms such as atmospheric deposition and water flow (Evangeliou et al., 2020). For Belgium, natural soil areas have not been studied yet. Especially for Flanders we can however expect still high concentrations due to our high degree of urbanization and industrialization.

Sediments are increasingly recognized as a significant sink for MPs in freshwater systems (Semmouri et al., 2023, Rodrigues et al., 2018). Due to their density and composition, MPs can accumulate in riverbeds, lake sediments, and estuarine environments, where they may persist for extended periods. Hydrodynamic factors such as water flow, turbulence, and sediment grain size influence the deposition and resuspension of MPs, leading to variations in their distribution (Guo et al., 2024a).

The understanding of microplastic pollution in soil is fragmented due the lack of harmonization and standardization in the quantification and methodologies used for sample collection, processing and analysis (Yang et al., 2021b). This methodological inconsistency hinders comprehensive comparisons of microplastic occurrence across different studies and research communities, but primarily also results in few studies being actually conducted. Only a handful of studies have studied the extent of microplastic contamination in soil and sediments. The global distribution of studies on MPs and microplastic concentration in soils is further explained in the next section.

3.2 What are the expected concentrations of microplastics in soils, sediments and groundwater?

3.2.1 Microplastic concentrations in soil

Microplastic contamination in soil is ubiquitous but varies temporarily and spatially (Li et al., 2019b). In this section of the report, we present a comprehensive review of 85 studies (published between 2018 and 2024, see Table S1 in Supplementary) with 199 investigated soils on microplastic contamination. Articles were acquired from several databases, including Google Scholar, PubMed and Web of Science. The keywords *microplastics** and *soils** were used to search these databases.

The majority of studies on MP contamination in soils (56.5%) have been conducted in Asia (Figure 9), with China as the main contributor. Europe accounted for 27.1% of these studies, conducted in The Netherlands, Germany, Portugal, France, Spain, Switzerland, Sweden, Denmark, United Kingdom, and Greece. North America accounts for 7.1%, Africa for 4.7% and South America for 2.4% of the studies (Figure 9). Fewer than 1.2% of the studies were conducted in Antarctica and Australia.

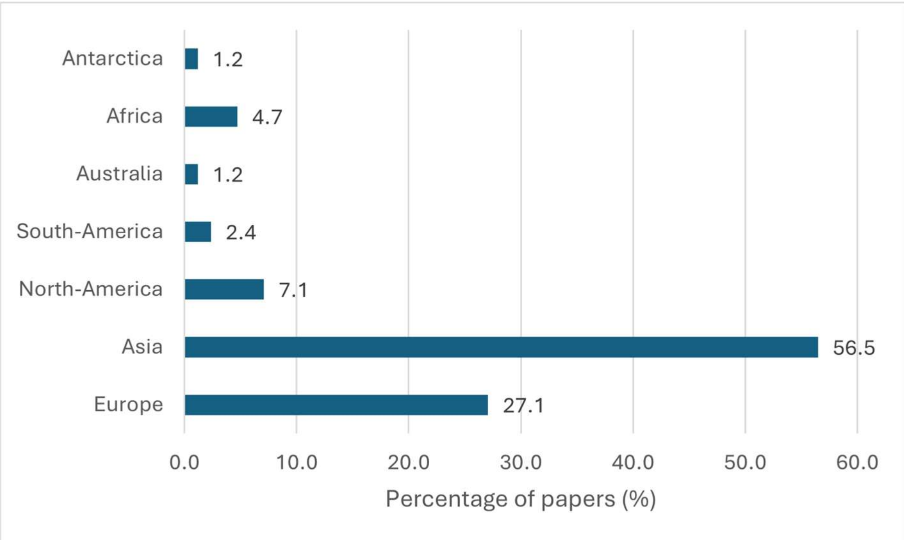

Figure 9 | Worldwide distribution of studies conducted on microplastic concentrations in soil. Papers selected from 2018 until 2024 (n = 85).

When examining the land-use types studied, more than half of the research was focused on agricultural soils, which refers to land used to produce crops, livestock, or both, accounting for 56.3% (Figure 10). The second prominent land-use type investigated is urban soils (15.1%), including industrial areas, roads, parking lots, city areas, dump sites and playgrounds. Other land-uses related to agriculture, such as plantations, horticultural soils, greenhouse soils and grasslands, accounted for respectively 2.0%, 4.5%, 3.5% and 4.0% (Figure 10). Natural ecosystems including forests (4.5%) and wetlands (3.5%), but also soils (coal mines, Antarctic soils, seagrass soil and savannas) in the category "Others" are less frequently sampled.

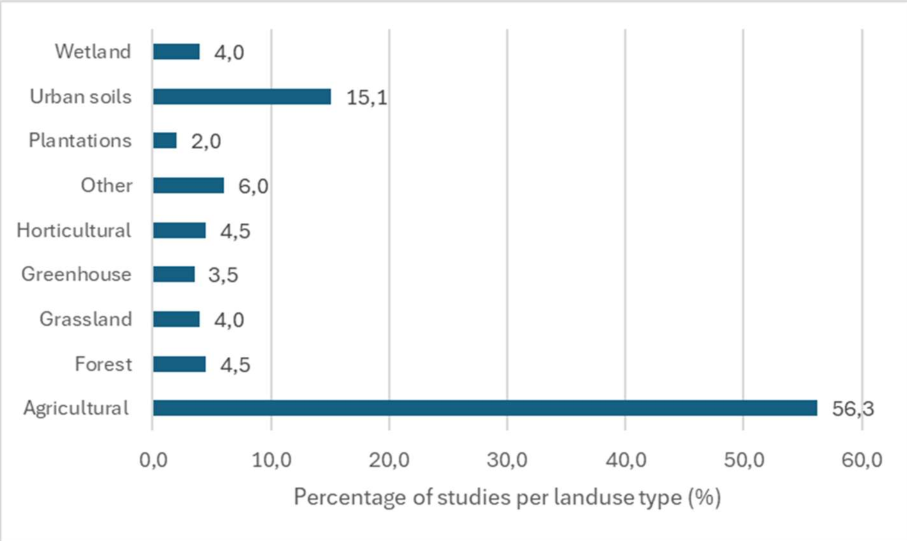

Figure 10 | The distribution of land-use types in the selected studies. The percentage of the types of land-use which are studied in the selected papers in this report (n soils = 199; n publications = 85).

The concentrations of MPs in soil vary considerably. Half of the reviewed studies (51.1%) reported an average concentration below 1 000 MP particles/kg soil (Figure 11). A significant portion of the studies (36.7%) report average concentrations between 1 000 to 10 000 MP particles/kg soil, while 12.3% of the studies reported average concentrations above 10 000 MP particles/kg soil.

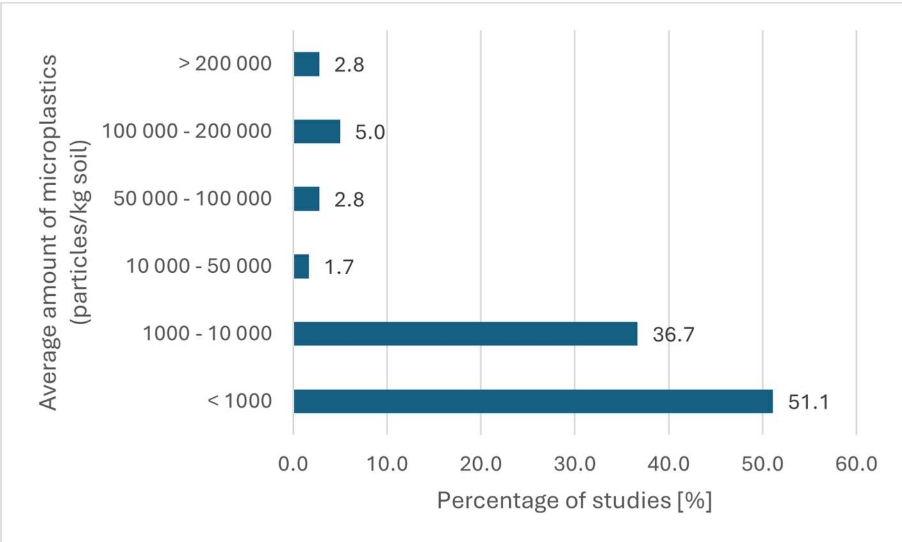

Figure 11 | The average amount of microplastics in soil within the investigated soils from the selected studies. The average concentration of microplastics grouped into six categories (< 1000, 10 000 – 50 000, 50 000 – 100 000, 100 000 – 200 000, > 200 000 particles/kg soil) (n publications = 74; n soils = 180).

Variations in microplastic concentrations are observed across different land-use types (**Table 6**). In agricultural soils, concentrations range from non-detectable levels to up to one million particles/ kg soil. The highest recorded microplastic concentration, measured at 12.7 million particles/ kg soil, was reported in Austria (Meixner et al., 2020).

Table 6 | Minimum and maximum abundance of microplastics per land-use type. The abundance of microplastics per land-use type (particles/kg soil)per land-use type. The maximum reported in a paper considering that land-use type is given. Number of papers considered per land-use type: agricultural (n = 97), farmland (n = 15), forest (n = 9), grassland (n = 8), greenhouse (n = 7), horticulture (n = 9), plantations (n = 4), urban soils (n = 30), wetlands (n = 8) and other (n = 12). Total number of soils considered (n soils = 199; n publications =85) .

Land-use type	Abundance (particles/kg soil)	
	Minimum	Maximum
Agricultural	0,34	843 808
Farmland	2	4 496
Forest	1,7	393 000
Grassland	2	92 000
Greenhouse	75,7	1 900
Horticultural	23	2 600
Other	1	12 700 000
Plantations	109	2 000
Urban soils	500	158 000
Wetland	666,1	99 000

These results should be interpreted with caution, due to variable quality in sampling, detection, quantification, and reporting methods amongst studies. A major challenge in microplastic research is the insufficient characterization of detected particles. Many studies fail to report key details such as size distribution and shape, despite evidence that smaller MPs have a greater impact on soil organisms (Redondo-Hasselerharm et al., 2024). Without this information, it is difficult to assess the ecological risks associated with diverse types of MPs.

In addition to detection challenges, the way microplastic concentrations are reported introduces further ambiguities. **Most studies provide only the average concentration without including raw data, limiting the reproducibility and comparability of results**. Additionally, reported concentrations are often expressed in particles/kg soil, yet it remains unclear whether this refers to dry or wet soil. These inconsistencies in detection and reporting significantly impact data interpretation and hinder cross-study comparisons. Given these concerns, it is crucial that future studies adopt **standardized protocols** to improve data quality and transparency.

Microplastic concentrations can differ significantly based on the entry pathways, land uses and vicinities (Büks and Kaupenjohann, 2020). Most studies focus on the difference between soils subject to one specific MP entry pathway (e.g. compost addition, sewage sludge, plastic mulch on agricultural and horticultural sites, sites near cities and on the countryside) and soils not subject to this pathway. As shown previously, microplastic research in soil is focused on agriculture (Figure 12). Research regarding industrial and natural areas, the input of MPs with road dust, littering, irrigation water, compost and digestates, are underrepresented or lacking. Therefore, there are no microplastic baseline concentrations that could be used for ecological risk assessment.

In conclusion, more research regarding microplastic concentrations in all kinds of soil environments is needed as well as standardized methods for extracting, identification and quantification of MPs, focusing not only on concentrations but taking into account also crucial parameters such as polymer size, shape and type.

3.2.2 Microplastic concentrations in groundwater

Groundwater is a vital resource important for various aspects of the environment and human life (Lee et al., 2024, Thomas et al., 2022). It is a crucial drinking water source for millions worldwide. Based on 58 studies in 25 countries, the global microplastic concentration in groundwater ranges between 0 up to 6 832 particles/L (Figure 12 and 13)(Xu et al., 2024).

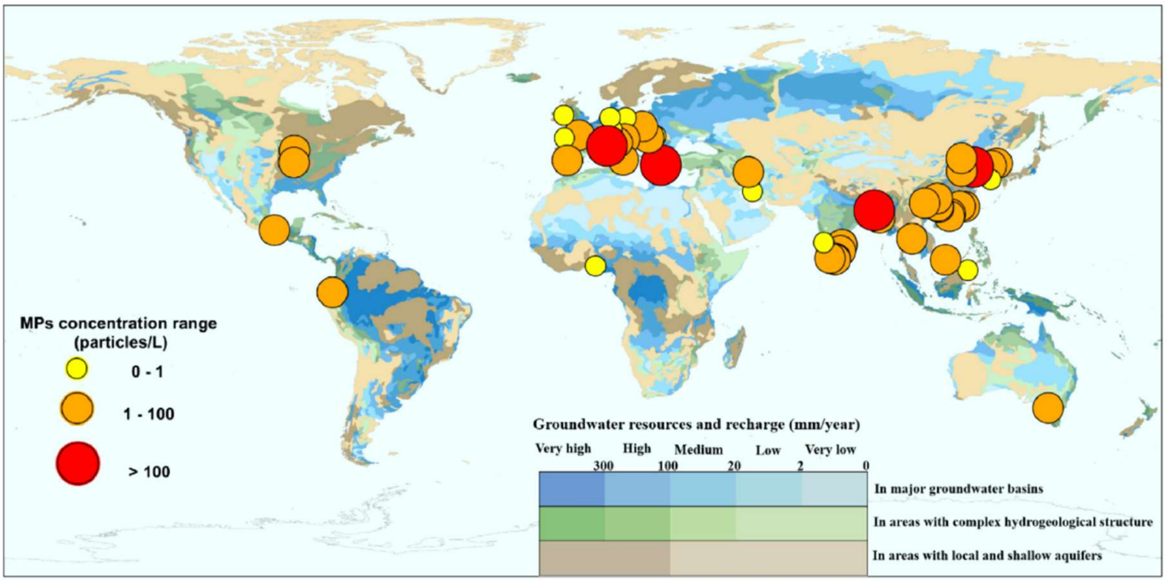

Figure 12 | Global distribution of the average MP concentration in groundwater. Source Xu et al. (2024).

An average concentration of 1 up to 100 particles/L is reported in most studies (Figure 13). A few field studies reported values below 1 particle/L and only four field studies report average microplastic concentrations exceeding the 100 particles/L. The lowest average microplastic concentration in groundwater ($7x10^{-4}$ particles/L) was detected from groundwater wells in Holdorf, Germany (Mintenig et al., 2019). The highest average abundance was measured in Jiadong Peninsula, China: 2 103 particles/L and a maximum of 6 832 particles/L (Mu et al., 2022).

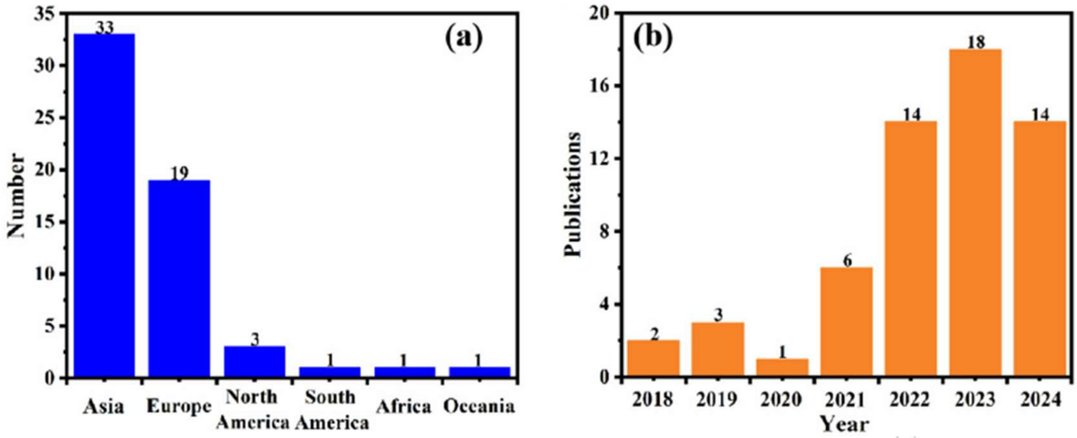

Figure 13 | Number of refereed articles on groundwater microplastics. (a) Number of refereed articles on groundwater microplastics sorted per continent (n = 58) and (b) Number of refereed articles on groundwater microplastics sorted per publication year (n = 58). Source Xu et al. (2024).

Similarly as for microplastic studies in soil, most studies on microplastic concentrations in groundwater are conducted in Asia (Figure 13) (Xu et al., 2024). Still, identification and quantification of MPs in groundwater is a relatively new field as the gross amount of the papers were published in the last four years.
As with the research on MPs in soil, similar deficiencies came up in the research on MPs in groundwater. The lack of universal and standardized methods for sample collection (volume), procedures, and on-site pretreatment methods specifically tailored for microplastic analysis in groundwater contributes to the limited research interest and amount of available data (Viaroli et al., 2022, Lee et al., 2022). Therefore, field studies on groundwater remain limited (Lee et al., 2024). In addition, groundwater sampling is less straightforward than soil sampling or surface water sampling, as a well and/or bailer (cylindrical tool to collect groundwater) need to be present. Similarly as for soil systems, reviewing data on microplastic concentration in groundwater is challenging as no standardized method is present.

In conclusion, the need for a standardized method for procedures and methods on collecting and analyzing MPs in groundwater is necessary (Lee et al., 2024). The average amount of MPs in groundwater varies significantly with values in between 0 and 6 832 particles/L.

3.2.3 Microplastic concentrations in sediment

The US Army Corps of Engineers (USACE) has made a public available online microplastic database for sediments (MP-SED, 2023) which contains a searchable database of MP concentrations, sizes, shapes and polymer data for sediments across diverse geographic locations (Wilkens et al., 2024). The online database includes papers from 2004 until 2023. Articles were acquired from several databases, including Google Scholar and Web of Science. The keywords *microplastics dredging dredge** and *microplastic sediments** were used to search these databases. The results focusing on sediment of rivers, estuaries and lakes are discussed.

As with soil and groundwater microplastic research, the majority of research on MPs in sediments is conducted in Asia (32%) (Wilkens et al., 2024). Asia is followed by Europe (31%) and North America (30%), with minor contributions from Africa and Oceania. The sampling locations were categorized into systems including lakes, rivers, the Great Lakes, estuarine and marine environments, harbors and ports and studies specifically addressing dredging (Figure 14). The median of the microplastic concentration is given in items/ kg dry sediment.

The results for Lakes and Reservoirs is composed out of 42 sampling sites from eight publications across 14 lakes. The median MP concentration was 184 MP particles/kg with a range of 9 to 5 450 MP particles/kg dry sediment. The highest concentration (5 450 MP particles/ kg dry sediment) in lakes was found in the rural head water of Muskoka-Haliburton, Ontario, where fibers were the most prevalent type, which suggests atmospheric deposition as the significant source (Welsh et al. 2022a; 2022b). Conversely, sediment from an urban lake in the UK was investigated and exhibited some of the lowest MP concentrations (Vaughan et al., 2017).

All studies (n = 4) conducted in the Great Lakes (Lake Michigan, Lake Erie, Lake Huron and Lake Ontario) found MPs (Wilkens et al., 2024). The median MP concentration was 290 MP particles/ kg dry sediment, with a range from 10 to 27 830 MP particles/kg dry sediment (Figure 14). The data of Lakes & Reservoirs and the Great Lakes reveals a substantial range in MP concentrations across urban and rural lakes, which suggest the complex presence and distribution of MP is influenced by varied factors such as atmospheric deposition, river connectivity, multiple sources and transport mechanisms beyond the human proximity (Wilkens et al., 2024).

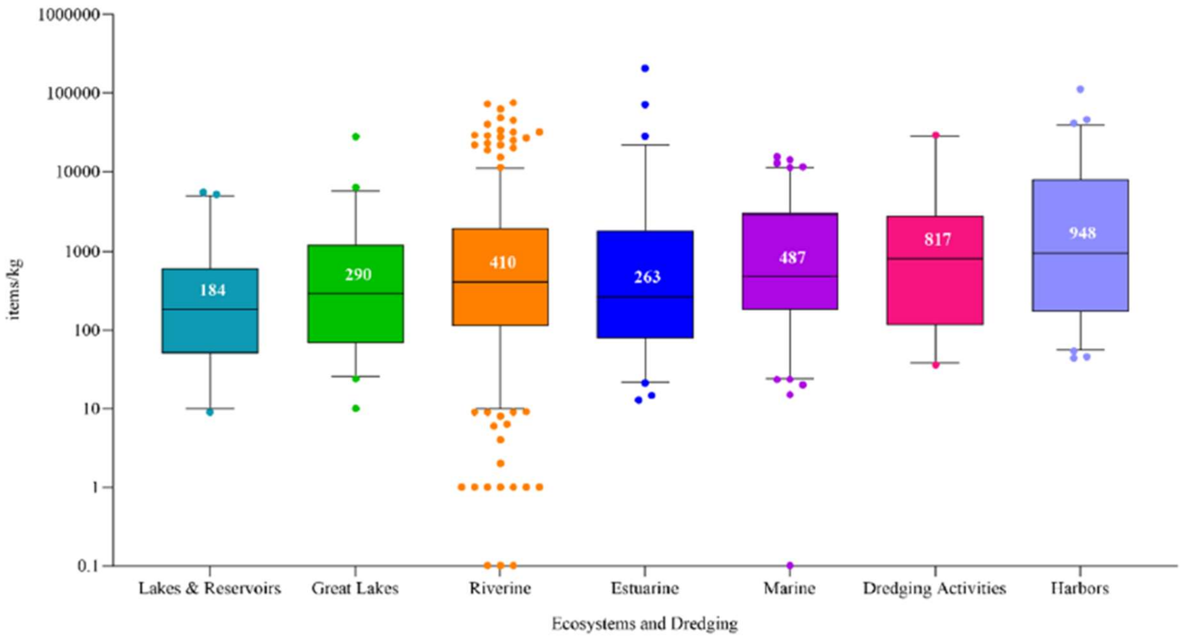

Figure 14 **| Distribution of microplastic concentrations (items/ kg dry sediments) in sediment samples. The box range = 25th to the 75th percentiles; median = horizontal line; whiskers denote the range of the 5th to the 95th percentiles. Dots indicate concentrations greater than 95th percentiles. (n researches = 122). Source Wilkens et al. (2024)**

Thirty papers in the MP-SED 2023 database collecting 429 surficial sediment samples across 75 rivers, all contained MP except for three samples (Wilkens et al., 2024). The MP concentrations range from 0 to 74 800 MP particles/kg dry sediment with a median of 410 MP particles/kg dry sediment (Figure 14). The highest average concentration of 32 947 MP particles/kg dry sediment (range 18 690 to 74 800 MP particles/kg dry sediment) was found in the Wen-Rui Tang River, China, an area with high industrial activity (Z. Wang et al. 2018). In the UK, MP concentrations of 31 950 MP particles/kg dry sediment (range 1700 to 62 200 MP particles/kg dry sediment) are reported in the River Glossop Brook and 21 300 MP particles/kg dry sediment (range 500 to 72 400 MP particles/kg dry sediment) in the River Tame (Hurley and Nizzetto, 2018). In this study, a decline in the MP concentration of 28 out of 40 sampling sites was noted due to a major flood event (mean pre-flood 7 036 MP particles/kg dry sediment; post-flood 889 MP particles/kg dry sediment). This shows that MP contamination can substantially change following flooding (Wilkens et al., 2024). Similar results were found for MP concentrations after a flood in the Santa Cruz River near Tuscon, Arizona (Eppehimer et al., 2021).

Fifteen studies on estuarine areas with 17 sampling sites (n = 72 samples) are included in the MP-SED 2023 database (Wilkens et al. 2024). The overall median MP concentration was 263 MP particles/kg, ranging from 13 to 205 859 MP particles/kg (Figure 14).

In conclusion, MP are present in the sediment of different freshwater areas ranging from 0 to 74 800 MP particles/kg dry sediment. The high variability in MP concentrations highlights the critical need for proactive measures at the source to prevent plastic contamination (Wilkens et al., 2024).

4 MEASURING TECHNIQUES MICROPLASTICS

4.1 WHICH TECHNIQUES ARE AVAILABLE TO MEASURE MICROPLASTICS IN SOIL, GROUNDWATER AND SEDIMENTS?

4.1.1 Conventional techniques for MP detection

The extraction and identification of MP in soil and sediment samples typically include four steps: (1) pre-treatment/drying, (2) purification, (3) MP extraction and filtration and (4) identification and quantification of MP. These steps can be carried out in variable order and/or repeated several times depending on the complexity of the sample (Figure 15):

- **Pre-treatments/drying**: Soil samples are prepared for sieving by air-drying, oven-drying or freeze-drying, as wet soil is harder to pass through a mesh. Sieving is performed to remove macroscopic plant residues, roots, and stones. The mesh sizes can vary between 1mm, 2mm and 5 mm, allowing for the manual selection of larger microplastic pieces for direct counting and identification.
- **Purification:** Soil organic matter can adsorb or encapsulate MPs, thus complicating separation from the soil matrix, and later visualization and quantification. Therefore, 30% H_2O_2 or Fenton's reagent are often used for natural organic matter digestion.
- **MP extraction and filtration:** MP detachment from the soil matrix is usually done by density separation in combination with stirring, centrifugation, or ultrasonic treatment. The released MP are typically suspended in the supernatant and collected by filtration through a membrane filter.
- **Identification and quantification:** Identification and quantification of MP is almost always done by visual inspection, even if followed by chemical characterization. From the reviewed studies by (Prata et al., 2019) on water and sediment (N = 40), 50% used Fourier-transform infrared spectroscopy (FTIR) in combination with visual inspection, 32.5% visual inspection only, 10% Raman spectroscopy in combination with visual inspection, whereas electron microscopy, staining dyes and gas chromatography-mass spectroscopy were each used for 2.5%.

- **FTIR spectroscopy** is one of the most commonly used analytical methods in microplastic research, as it enables precise identification of polymer types. Its ability to determine abundance, shape, and size is somewhat limited (Shim et al., 2017). Recent advancements in micro-FTIR (μ-FTIR) imaging have enabled automatic identification of MP concentrated on the filter membrane without pre-sorting (Vianello et al., 2019). FTIR spectra of unknown plastics are identified by comparing the recorded spectrum to a reference library, though assigning unidentified spectra to specific chemical species remains challenging and depends on expert interpretation or the availability of suitable reference libraries (Chen et al., 2020b).
- **Visual inspection** allows the classification of polymer particles based on their physical characteristics, observed directly, or using a stereoscope or microscope. This is one of the most widely used methods of identification and quantification of polymer particles and is often employed as a pre-selection step before chemical characterization (Prata et al., 2019). However, this approach is highly prone to bias, with error rates ranging from 20% to 70% (Möller et al., 2020). In the past, studies recommend the "hot needle test", which leverages the thermoplastic properties of many synthetic polymers to distinguish plastics from natural particles, but does lack precision (Galgani et al., 2013). Scanning electron microscopy (SEM) can provide extremely clear and high-magnification images of plastic-like particles (Cooper and Corcoran, 2010). Additionally, high-resolution imaging of surface textures can aid in discrimination, while energy-dispersive X-ray spectroscopy provides elemental composition analysis of the same object (Vianello et al., 2013).
- **Raman spectroscopy** has also been used to identify MPs. The laser beam falling on an object results in different frequencies of back-scattered light, depending on the molecular structure and atoms present, which produce a unique spectrum for each polymer. Raman analysis not only identifies plastics, but also provides profiles of the polymer composition of each sample similar to FTIR (Kappler et al., 2016). In terms of the combination of non-destructive chemical analysis with microscopy, Raman spectroscopy is comparable to the FTIR method, including the requirement for expensive instrumentation.
- **Staining dyes** (e.g., Nile red staining) provide an alternative and complementary method to help distinguish polymer particles from other materials, reducing the risk of missing small or transparent plastics that are difficult to identify manually (Liu et al., 2024).

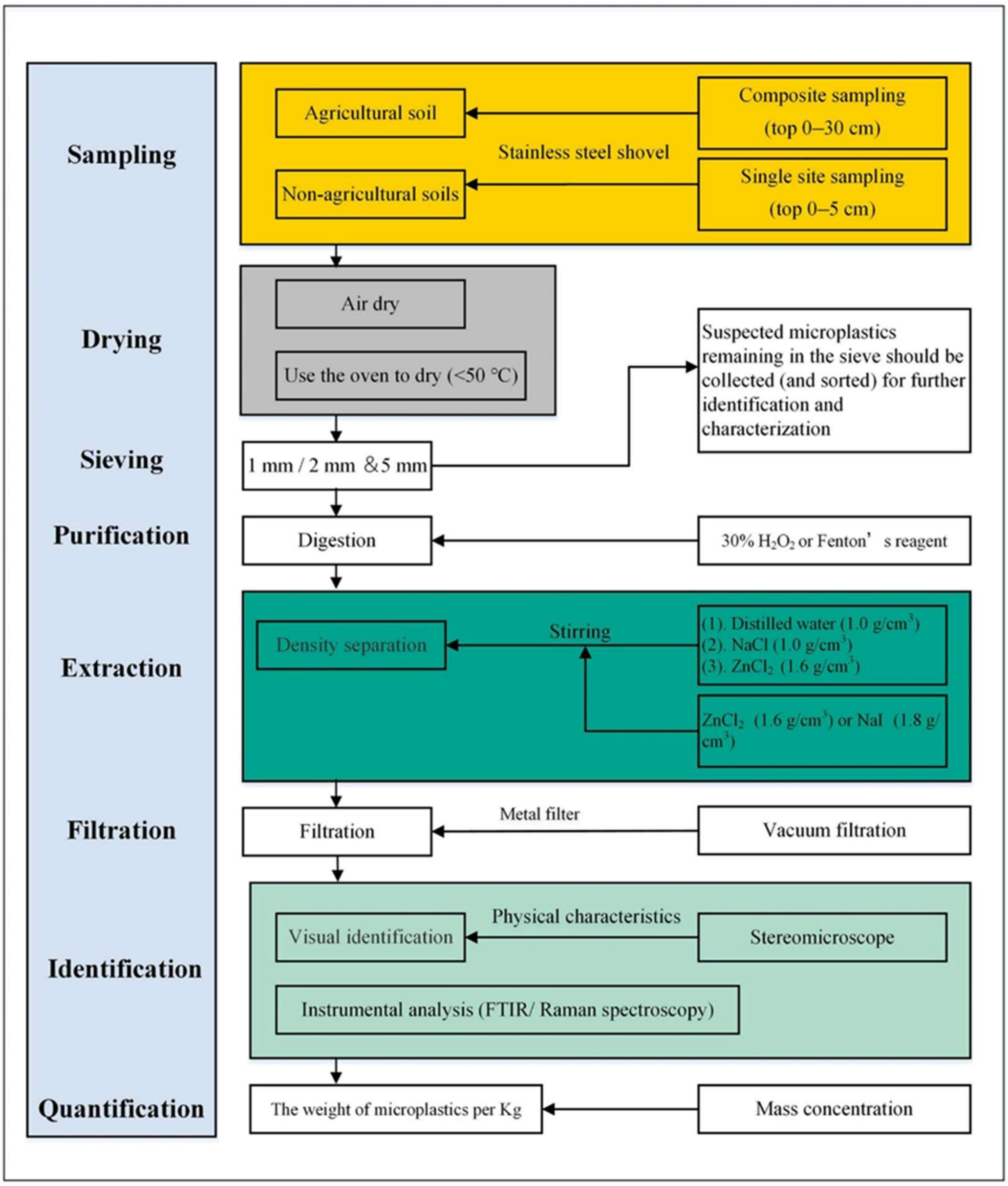

Figure 15 | Schematic overview of conventional MP detection techniques in soil. Source (Yang et al., 2021b).

4.1.2 Optimized techniques for MP detection

To improve the accuracy and efficiency of MP detection in soils and sediments, current optimization efforts go in two main directions: (1) improving detection efficiency and (2) reducing the detection limit for greater precision. These improvements target three key problems in conventional MP detection: more efficient/complete removal of organic matter, more efficient extraction of MP, and more accurate identification of MP. The methods for MP detection at different stages can be combined and applied to MP measurement for various purposes and environments. Below we first provide an overview of options for optimization of MP extraction and further elaborate on some of these methods in Chapter 4.2.

4.1.3 Optimized purification

There is a need to create a simple method of digestion capable of reducing organic matter without affecting the structural or chemical integrity of polymers (Felsing et al., 2018). Nowadays, oxidation through the use of 30% H_2O_2, Fenton's reagent or a combination of both is mostly used. Besides oxidizing methods digestion can also be acidic (5–69% HNO_3, 5–37% HCl), alkaline (NaOH, and KOH) or enzymatic (Tripsin, Collagenase and Papain) (Prata et al., 2019). For optimized purification methods, acidic and alkaline digestion methods are frequently reported in literature to remove the organic fraction of the sample matrix. Enzymatic treatment presents a promising alternative to chemical methods, as it is less harmful to the environment and utilizes biologically active enzymes for improved separation.

4.1.4 Optimized MP extraction

MP extraction is mostly done making use of density separation, including NaOH, NaI or NaBr to the medium. Some advancements in MP separation aim to overcome existing limitations, primarily focusing on this density separation (Liu et al., 2019) or plastic flotation and separator system based on density separation (Li et al., 2024). Alternatively, electrostatic separation harnesses the electrostatic properties of polymer particles in separation efforts (Felsing et al., 2018), while magnetic extraction of MP relies on attaching magnetic nanoparticles (MNS) to the MP surface, followed by MP separation by applying magnetic force (Liu, Under review). In the section Advantages and disadvantages, these methods will be further elaborated.

4.1.5 Optimized identification and quantification

Microscopy, µ-FTIR, Raman spectroscopy, dye technology, gas chromatography-mass spectrometry (GC-MS) and pyrolysis GC-MS based detection are by far the most frequently used methods for microplastic detection (Hermabessiere et al., 2018, Santos et al., 2023). However, alternatives to these methods are being investigated, including surface-enhanced Raman spectroscopy (SERS), laser direct infrared spectrometry (LDIR) and infrared photothermal heterodyne imaging analysis (IR-PHI) (Dey, 2022, Luo et al., 2022, Kniazev et al., 2021). It is beyond the scope of this review to go into detail on all these methods, but they each come with their specific advantages and limitations, and most if not all are still in the research phase and undergoing further development, and are still far from a stage to be implemented in routine detection of MP.

The rapid developments in this very recent research field put further challenges to standardization, because new methods are continuously being developed before existing methods have been thoroughly tested and validated, let alone standardized.

4.2 WHAT ARE THE ADVANTAGES AND DISADVANTAGES OF EACH TECHNIQUE?

In this section, the advantages and disadvantages of each method are discussed separately based on purification, extraction, and identification/detection techniques (**Table 7**).

Table 7 | A summary of the advantages and disadvantages of technologies for soil microplastics detection.

Technology	Isolation and detection	Advantages	Disadvantages	References
Purification	Conventional 30% H_2O_2 or Fenton's reagent	Easy to operate Cost-effective	MP fragmentation or degradation	(Scheurer and Bigalke, 2018)
	Acidic/Alkaline digestion	Suitable to remove organic fraction of the sample matrix	Some MP can degrade Time-consuming	(Möller et al., 2020)
	Enzymatic digestion	Does not degrade MP if properly selected Environmentally friendly.	Selecting the appropriate enzyme Time-consuming Costly	(Möller et al., 2020, Zhu and Wang, 2020)
Extraction	Conventional density separation	Easy to operate Cost-effective	Cannot extract polymers with densities higher than the flotation solution Soil adsorption competition Background interference Human errors Time-consuming	(Möller et al., 2020)
	Plastic flotation and separator system	Avoid transfer loss of small particles	Soil adsorption competition soil adsorption Background interference Human errors	(Li et al., 2024)
	Density separation-Circulation of NaBr solutions	Can process large sample numbers	Soil adsorption competition Background interference Human errors	(Liu et al., 2019)
	Olive oil separation	Easy to operate Inexpensive Isolate various polymers	Soil adsorption competition Change MP surface properties Background interference Human errors	(Scopetani et al., 2020)
	Accelerated solvent extraction	High automation Low cost	Soil adsorption competition Background interference Human errors	(Fuller and Gautam, 2016)
	Magnetic extraction	Low size limit High accuracy Time-saving Easy to operate Inexpensive	Soil co-labelling Background interference Human errors	(Liu et al., 2024)
	Electrostatic separation	Simplifies treatment and preparation of field samples; Time-saving.	Further validation is needed for its size limit Loss of small-sized MP Only fits for dry samples	(Felsing et al., 2018)
Identification and quantification	μ-FTIR	Providing information on the specific bonds of plastics Easy to operate Inexpensive	Low accuracy High detection limit (10 μm).	(Shim et al., 2017, Löder et al., 2015)
	Microscopy	Easy to operate for counting Inexpensive	No plastic chemical information Low accuracy Time-consuming	(Möller et al., 2020)
	Dye technology	Promote MP visualization	Overestimation of the MP concentration.	(Liu et al., 2024)

		Information on MP number, size, and shape; Semi-automatic detection	Background interference; Changes of MP characters interference	
	Raman spectroscopy	Providing information on the specific bonds of plastics Additional information on fillers and pigments Low detection limit (5 µm)	Low accuracy, especially paint particles. Week signal for MP< 5 µm	(Kappler et al., 2016)
	Surface-enhanced Raman spectroscopy	Providing information on the specific bonds of plastics; Lower detection limit compared with Raman spectroscopy; Time-saving; Easy to operate; in-situ analyte identification.	Requirement of substrate analyte close contact Substrate degradation Selectivity issues Problems with reusability and homogeneity of substrate	(Dey, 2022)
	Laser Direct Infrared Spectrometry (LDIR)	Rapid analysis of MP > 10 µm Identify MNP types and morphology Easy to operate.	Relatively large detection limit Costly	(Nizamali et al., 2023)
	Infrared Photothermal Heterodyne Imaging Analysis (IR-PHI)	Identification of MNP type and morphology Quantify MNP in a single highly sensitive analysis Small detection limits (200 nm)	Very low sample throughput	(Kniazev et al., 2021)
	pyrolysis-GC/MS; TED-GC-MS	Relatively fast Easy to operate	Background interference No information on MP number, size, and shape Costly	(Dumichen et al., 2017)

4.2.1 Purification techniques

The conventional purification methods (30% H_2O_2 or Fenton's reagent) for MP are relatively simple to operate, do not require major investments and can thus be done in most environmental laboratories, provided that measures are taken to minimize sample contamination. However, a fundamental problem is that the repeated purification and extraction steps are quite drastic, potentially altering the chemical composition and/or leading to further fragmentation or degradation (Scheurer and Bigalke, 2018). Moreover, samples with high organic matter content require long digestion times, as organic matter can adsorb or encapsulate MP and significantly reduce the detection efficiency.

The optimized acidic digestion was found to efficiently remove biologic material by optimizing concentration and temperature, ensuring effective removal within a reasonable timeframe. However, some polymers (e.g. nylon, PET – polyethylene terephthalate) have low resistance to acids and may also be degraded, especially at high concentration of acids and elevated temperatures (Liu et al., 2024). Alkaline digestion (NaOH) was found to destroy PA and PE fibers while leading to melting or discoloration of other polymers (Covernton et al., 2019). Foekema et al. (2013) digested fish intestines with 10% KOH solution at room temperature for 2–3 weeks. While apparently successful and non-destructive to synthetic polymers, the procedure is exceedingly time-consuming, and thus not suitable for routine measurements, and may not be applicable for plant material or stabilized soil organic matter.

In general, purification of samples with strong acidic or alkaline solutions will lead to uncontrolled alterations to the microplastic composition of the sample. Enzymatic treatment offers an eco-friendly alternative to chemical methods, leveraging biologically active enzymes for better separation. The enzymatic method was compared to digestion protocols using HCl and NaOH, demonstrating the highest efficiency while also offering the advantage of not degrading the polymer particles. However, despite its advantages in MP separation, challenges remain in selecting the appropriate enzyme for specific samples and adapting to complex treatment protocols (Zhu and Wang, 2020). Furthermore, the use of enzymes is costly, creating a significant economic barrier to routine implementation.

4.2.2 Extraction techniques

For the separation of MP from soil/sediment, density separation protocols are the most commonly applied using high density salt solutions as extraction media (Möller et al., 2020). A saturated NaCl solution has a maximum density (ρ) of 1.2 g/cm³ and cannot extract synthetic polymers with higher densities, such as PET and PVC. Alternative solutions such as NaI (ρ = 1.8 g/cm³), $Na_6[H_2W_{12}O_6]$ (ρ = 1.4 g/cm³), Zn_2Cl (ρ = 1.6–1.7 g/cm³), and NaBr (ρ = 1.55 g/cm³) are recommended, but their cost and hazardous nature may limit their application (Möller et al., 2020, Luo et al., 2022).

Advancements in MP separation aim to overcome existing limitations of strong interaction between native soil organic matter and MP, primarily focusing on optimized density separation, e.g. density separation-circulation of NaBr solutions (Liu et al., 2019), or using a plastic flotation and separator system involving $ZnCl_2$ solutions (Li et al., 2024). Yet, the strong interactions between small-sized MP and soil particles render separation methods only based on density differences between MP and soil particles ineffective for small MP. An alternative separation method that has been explored is electrostatic separation, which harnesses the electrostatic properties of MP. However, the soil must be oven dried (105°C) before the electrostatic separation can be performed, and the reported sized limit of electrostatic separation methods remains large at 63 µm (Felsing et al., 2018). Recently, Liu (Under review) explored the potential for magnetic extraction of small MP (up to 4 µm), by attaching magnetic nanoparticles (MNS) to the MP surface, followed by MP separation by applying magnetic force. Although still in the research stage, this method appears to be promising for extraction of the smallest MP from soil.

4.2.3 Identification and quantification techniques

For **identification and quantification** of MPs, conventional MP size detection from soil and sediment samples is typically limited to 50 µm (with most studies focusing only on the large MP fraction of several hundreds of µm), and very few have examined particles as small as 10 µm (Liu et al., 2024, Zhou et al., 2021), leaving the smallest fractions with presumably the largest environmental impact largely undetected (Covernton et al., 2019). µ-FTIR is the most frequently used conventional technique in diagnostic analysis of plastic polymers by providing information on the specific bonds of plastics. In theory, µ-FTIR can detect MPs as small as 10 µm in diameter (Shim et al., 2017). The µ-FTIR imaging equipped with focal plane array (FPA) detectors facilitates a much faster generation of chemical map of MP by simultaneously records several thousand spectra within one single measurement (Vianello et al., 2019). The first study utilizing FPA-based µ-FTIR imaging to analyze MP in environmental samples was reported in 2015, demonstrating the rapid detection of MP as small as 20 µm with high lateral resolution (Löder et al., 2015).

In addition, dye technology is an alternative method to enhance MP visualization in combination with microscopy, providing information on their number, size, and shape. For this, Nile red is an often use. However, dye technology have the drawback that they can lead to overestimation of MP concentration due to co-staining of natural organic matter and interference from background changes in stained MP characteristics (e.g. chemical characteristic spectra and color), and should therefore handle with care.

A third often used method is Raman spectroscopy. Raman spectroscopy has a lower detection limit, with MPs as small as 5 µm identified in practice, compared to FTIR (Kappler et al., 2016). Raman microspectroscopy can provide additional information about contained fillers and pigments, which are mostly not available by FTIR microspectroscopy. However, in some cases, solely using Raman microspectroscopy can lead to misidentification, especially of paint particles (Kappler et al., 2016). Additionally, for MP smaller than 5 µm, a drawback is that Raman signals can be weak. To overcome this, one can employ Surface Enhanced Raman Spectroscopy (SERS), which can reduce the MP size detection limit. The advantages of SERS are high-resolution sharp peaks (in comparison to fluorescence) aiding simultaneous multi-component analysis, speed of analysis, in-situ analyte identification and portability of the instrument. However, in SERS, the substrate plays a crucial role in enhancing the MP Raman signal of analyte molecules. The substrate, typically made of nanostructured noble metals (e.g., gold, silver, or copper), generates localized surface plasmon resonances (LSPRs) when excited by incident light. The main limitations of SERS are requirement of substrate-analyte to prevent melting and substrate degradation mainly caused by Raman laser, selectivity issues, and problems with re-usability and homogeneity of substrate.

All previous detection technologies have the drawback that they are less suitable to measure small size MPs and e.g. fail to detect tyre wear particles, the presumed most important source of environmental MP. Possible chemical analytical methods to detection small MP are thermoanalytical methods like py-GC/MS. Tyre wear particles can be measured though by pyrolysis GC-MS, which also has low (but MP type dependent) detection limits, e.g. 0.1 µg for PU (Santos et al., 2023). Pyrolysis GC-MS is however relatively costly and by definition does not provide any information on important physical MP characteristics such as size, morphology or number of

MP (Hermabessiere et al., 2018). Other emerging technologies for MP detection include full infrared photothermal heterodyne imagining (IR-PHI). Full infrared photothermal heterodyne imagining (IR-PHI) analysis allows to identify MP types, morphology and numbers in a single, highly sensitive analysis with extremely low detection limits (200 nm) (Kniazev et al., 2021). However, IR-PHI analysis has a very low sample throughput (Kniazev et al., 2021).

4.3 QUALITY ASSURANCE/QUALITY CONTROL

To date, no standardized method to extract, detect, identify and quantify MPs in soils exists (Meixner et al., 2020). However, the need for a reliable standardized method for sampling, sample extraction and quantitative measures is mandatory to compare data and legislative work. Furthermore, standardization in positive and negative controls to evaluate the methods is necessary but not present (Razaviarani et al., 2024). The quantification of MP in environmental samples has not been fully optimized and faces several challenges, including contamination, overestimation, and underestimation. These issues become particularly evident during the extraction process, where validation studies and blanks must be incorporated as essential components of the analytical procedure (Nuelle et al., 2014).

Synthetic polymers are everywhere, posing an elevated risk of contamination during MP sampling and analysis, such as from plastic equipment, synthetic fibers from shoes and clothing, or airborne particles. Therefore, precautions should be taken at each step, replacing plastic materials with alternatives like metal or glass whenever possible (Möller et al., 2020). To monitor potential sample contaminations, it is essential to include blank samples that undergo the same treatment as the environmental samples, as well as monitoring used liquids and the ambient air (e.g., by laying out wet filter papers for a defined amount of time) (Woodall et al., 2015). Field blanks are prepared, treated, and transported alongside actual samples, often filled with a microplastic-free matrix like kiln-treated sand to mimic sample conditions (Carter and Gregorich, 2007). The need for robust quality control in MP studies to also ensure reliable results. Quality control involves standardized procedures such as instrument calibration, use of reference materials, and strict protocols for sample handling, which help identify and minimize errors. It also enables data comparison over time and across locations to identify trends. Although this may seem very logical, many published studies, especially in the first years of soil MP research, did not adhere to these good practices, thus producing unreliable results.

For each environmental type described above, the lack of standardization is one of the main bottlenecks in microplastic quantification. Quality Assurance and Quality Control (QA/QC) criteria should be implemented in all future studies. Recently developed QA/QC guidelines for soil environments provide a framework for evaluating the quality of existing studies and serve as a protocol for best practices in future research on microplastic concentrations in soil (Table 8) (Redondo-Hasselerharm et al., 2024, Redondo-Hasselerharm et al., 2023).

Standardization efforts are already implemented for the microplastic detection in water samples (ISO norm ISO/DIS 16094-2) and a new ISO standard for environmental samples (ISO/FDIS 24,187) is currently in progress. Ensuring that a QA/QC framework is in place before conducting analyses is crucial for improving the reliability, comparability, and overall quality of microplastic research.

Table 8 | Guidelines for microplastic sampling, analysis, and identification. Summary of key considerations for sampling, handling, analysis, and identification of microplastics, including best practices to minimize contamination, ensure reproducibility, and validate analytical methods. Source (Redondo-Hasselerharm et al., 2024, Redondo-Hasselerharm et al., 2023).

Guidelines	Description
Sampling and storage recipient	Specify the tools used for sampling (e.g., stainless steel shovel, corer, spatula). Identify appropriate storage containers (e.g., glass jars, aluminium foil, metal containers) to prevent contamination.
Sampling Location	Provide detailed information on the sampling location (GPS coordinates, land use, environmental conditions). Follow a documented sampling protocol to ensure consistency.
Sampling Method and Depth	Describe how samples were taken, including the use of sub-replicates. Record the depth at which samples were collected (e.g., surface soil, different depth layers).
Sample size	Indicate the volume of the sample container used for collection.
Sampling date	Document the exact date and time of sampling.
Replicates	Specify the number of replicate samples collected per site to ensure reproducibility.
Field blanks	Use field blanks to assess contamination during sampling and transport.
Sample handling and storage	Detail how samples are processed and stored to prevent contamination (e.g., stored in dark, cool conditions, avoiding plastic exposure). Implement contamination-free sample transport (e.g., sealed containers, controlled environment).
Sample analysis	
Controls	Include negative controls (lab/procedural blanks) to assess background contamination. Use positive controls with known micropolymer particles to verify recovery efficiency.
Use of Plastic in the Lab	Minimize plastic use to avoid contamination. Record and justify any unavoidable plastic use.
Rinsing/Cleaning equipment and work area	Use filtered water and organic solvents (if applicable) to rinse equipment. Clean surfaces before and after use to prevent cross-contamination.
Non-synthetic clothing	Wear cotton or other non-synthetic lab coats and gloves to reduce fiber contamination.
Clean air	Work in a controlled environment with clean air supply (e.g., fume hood, laminar flow cabinet).
Keeping units covered	Cover all samples, filters, and solutions when not in use to prevent airborne contamination.
Method validation	Validate analytical methods for accuracy, precision, and reproducibility. Assess recovery rates using spiked samples with known MPs.
Identification of Polymer Type	Determine polymer composition using spectroscopic techniques (e.g., FTIR, Raman spectroscopy).
Total number of extracted suspected MPs reported	Report the total count of suspected MPs extracted.
Number/Portion of total extracted particles used for polymer identification	Specify the percentage or number of extracted particles that were analyzed for polymer composition.
False detection	Document false positives and measures taken to minimize misidentification.
Lowest detected particle size	Report the smallest particle size reliably detected by the method used.
Recovery efficiency assessment	Conduct recovery tests to determine the efficiency of microplastic extraction methods.
Quality Assurance of spectroscopic analysis	Verify polymer identification through reference spectra and replicate analyses.

4.4 WHAT ARE THE ANALYSIS COSTS AND HOW FAR ADVANCED ARE THE LABORATORIES IN FLANDERS IN THIS RESPECT?

Currently, the detection of MP in soil is primarily conducted by universities and research institutions. However, industrial laboratories (e.g. Eurofins, Measurlabs, and UKCEH's microplastic laboratory) are increasingly expanding into this field, offering specialized microplastic analysis services (Table 9). Eurofins conducts qualitative analysis to identify polymer types and uses spectroscopic techniques (Raman, FTIR, and LDIR) to quantify MPs.

Table 9 | Commercial laboratories offering microplastic analysis for soil and other matrixes

Company	Website
Eurofins	https://www.eurofins.be/nl/
Measurlabs	https://measurlabs.com/solutions/microplastics-testing/
UK Centre for Ecology	https://www.ceh.ac.uk/solutions/laboratory-services/microplastics-analysis

Given the enormous variety in methods for extraction and identification, the lack of standardization, and the continuous appearance of new methodologies, it is extremely challenging to provide general estimates for costs of MP analysis. As in all soil analyses, the most crucial step for MP detection is the sampling, which needs to be done in a representative manner adhering to good sampling practices. A critical point in MP sampling and analysis is avoiding unintended contamination, in particular for samples where MP concentrations are expected to be low, and/or when it is the ambition to extract and analyze the small MP fraction. So the numbers given here should be seen only as very approximate and will be greatly influenced by the factors mentioned above, including the need to include negative and positive controls.

The cost estimation provided here is based on "conventional" MP detection methods, and is for a hypothetical case of microplastic pollution assessment in a 5-hectare field with 18 samples in the Flanders region and includes the consumables, costs for use of analytical infrastructure, and labor costs for soil sampling, extraction and chemical characterization and quantification (Table 10). Additionally, the cost applied under the MiCoS project is displayed under Table 11.

Table 10 | The analysis costs for MP detection in soil for 18 samples including sampling, drying and sieving, MP extraction, identification, counting and data analysis.

Step	Details	Material cost	Equipment	Equipment cost	Time	Labour costs €50/h	Total
1. Sampling	Sampling 9 points with two soil layers (total 18 soil samples)	€20	/	/	6h	€300	€320
2.Drying and sieving	2 mm stainless steel mesh		/	/	4h	€200	€200
3.Extraction	5g dry soil with 30 ml Flotation Reagent (e.g. NaI, NaBr) and centrifuge, repeat 2 times	NaBr €20			16h	€800	€820
		NaI €60			16h	€800	€860
4.Purification	Digestion using reagent, repeat 2 times	30% H_2O_2 €20			16h	€800	€820
		NaOH €20			16h	€800	€820
5.Identification	Count MP amount and analysis particle chemical characterization		FTIR: €90/h	10 h: €900	10h	€500	€1400
			Raman: €125/h	16 h: €2000	16h	€800	€2800
6.Manual counting			Microscope	€50	8h	€400	€450
7.Data analysis and report writing					8h	€400	€400
Total for one field (18 soil samples)	FTIR: €4410 (NaBr), 4450 (NaI) Raman: €5810 (NaBr), 5850 (NaI)						
Total for one soil sample	FTIR: €245 (NaBr), €247 (NaI) Raman: €322 (NaBr), €325 (NaI)						
Notes	Simultaneous determination of soil 18 samples in the same batch.						

Table 11 | The cost for 10 samples using the MiCoS method, including sampling, drying, sieving, MP extraction, identification and data-analysis. Additional costs of equipment are not included.

Step	Details	General material cost	Time	Labour costs €50/h
Sampling	Ten soil samples	€50	6 h	€300
Drying and sieving	2 mm sieving	€20	3 h	€200
Extraction	10 g soil	€50	9 h	€600
	Digestion using 30% H2O2	€20	3 h	€150
	Digestion using Fenton's reagent	€20	3h	€200
Identification	Fluorescent microscopy with: Nile Red PTFE filters	€90	14h	€800
Data analysis and report writing			8h	€500
Total for 10 samples		€250	46h	€2300

Table 12 highlights the academic research groups in Flanders that are currently engaged in research on the distribution of microplastics (MPs) in field environments.

Table 12 | The teams currently publishing research related to field MP distribution.

Institution	Person in charge	Research matrix
Ghent University (UGent), Department of Biochemistry and Microbiology, K.L. Ledeganckstraat 35, Ghent, 9000, Belgium	Prof. Caroline De Tender	Soil
Ghent University (UGent) – Department of Environment, Faculty of Bioscience Engineering, Ghent University, 9000 Ghent., Belgium	Prof. Stefaan De Neve	Soil

5 ECOTOXICOLOGICAL AND HUMAN RISKS

5.1 WHAT ARE THE POTENTIAL RISKS OF MICRO- AND NANOPLASTICS ON PLANT, SOIL LIFE AND SOIL HEALTH?

The omnipresence of MNPs in the environment has emerged as a significant environmental concern, particularly regarding its impact on soil, sediment, and water bodies. These particles introduce a novel stressor that exerts an influence on soil properties, as well as soil fauna flora (En-Nejmy et al., 2024, Joos and De Tender, 2022). However, the precise mechanisms through which MNPs alter soil properties remain to be fully elucidated (En-Nejmy et al., 2024).

5.1.1 Impact on soil physicochemical properties

MNPs can be introduced into the soil through various processes including bioturbation, soil management practices, and water percolation (de Souza Machado et al., 2018b). Their presence has been shown to 1) decrease soil porosity and bulk density, 2) change water availability, 3) increase soil pH, and 4) increase dissolved organic matter (DOM) (En-Nejmy et al., 2024, de Souza Machado et al., 2018b, Joos and De Tender, 2022).

Soil porosity and bulk density are two important parameters of soil health, both of which have been demonstrated to be influenced by the presence of MPs. A study by Zhou et al. (2023) reported a decrease in both soil porosity and bulk density in soils amended with PE at concentrations ranging from 0.5% to 8% (w:w). Similarly, addition of MP types including PS, PP, PET, PES, and HDPE, at concentrations up to 2%, has been observed to result in a decrease in bulk density (de Souza Machado et al., 2018a, de Souza Machado et al., 2019, de Souza Machado et al., 2018b). Additionally, **MNP particles can modify soil structure** by occupying the interstitial spaces between soil particles, thereby interfering with natural pore spaces and aggregate formation (Joos and De Tender, 2022, de Souza Machado et al., 2018a, En-Nejmy et al., 2024, Zhang et al., 2019). Lehmann et al. (2019) reported that the addition of PE fiber at concentrations over 0.2% (w:w) reduces soil aggregate stability. Similarly, MNPs have been shown to weaken the forces that bind soil particles together (de Souza Machado et al., 2018b, Zhang et al., 2019). This phenomenon has been most notable in loamy-sand soil (Zhai et al., 2024).

Contradicting results have been reported regarding the impact of plastic fibers on water availability in soil (Joos and De Tender, 2022). Amending soil with PE fibers can increase its water availability (de Souza Machado et al., 2018a). Conversely, the addition of plastic film fragments of various sizes at concentrations of either 0,5% or 1% (w:w) led to higher rates of water evaporation and increased soil drying. This suggests that the addition of plastic to soil can negatively impact water retention, potentially leading to more rapid soil moisture loss (Wan et al., 2019). This phenomenon is further compounded by the reduction in infiltration rate, attributable to the occlusion of soil pores and interstitial spaces, resulting in diminished permeability and hydraulic conductivity (Ren et al., 2021, Zhai et al., 2024). Indeed, **changes in water holding capacity** have been observed in response to the addition of MNPs (En-Nejmy et al., 2024, Zhang et al., 2019).

The presence of MNPs in soil has also been demonstrated to **influence the soil's chemical properties**. Indeed, **changes in pH** upon addition of MPs have been reported. For instance, Boots et al (2019) demonstrated a decrease in pH upon addition of 0.1% (w:w) HDPE to soil (Boots et al., 2019). Another significant factor that can be impacted by MNP pollution is the **dissolved organic matter (DOM).** The decomposition of DOM plays a pivotal role in preserving soil fertility and structure (Zhai et al., 2024). MPs have adsorption capacity, enabling them to bind essential nutrients from the environment, thereby making them less available for surrounding microorganisms and plants (Brown et al., 2023b, Huang et al., 2022, Moreno-Jiménez et al., 2022, Shi et al., 2022a). The presence of MNP particles can act as an impediment, thereby **restricting the accessibility of DOM** to soil microorganisms and consequently impeding the rate of decomposition. This will eventually result in an accumulation of organic matter within the soil matrix (Zhai et al., 2024). Indeed, a study conducted on Chinese loess soil revealed that incorporating PP particles at very high concentrations led to an increase in soil enzyme activity, which consequently resulted in the accumulation of dissolved organic carbon, nitrogen and phosphorus (Liu et al., 2017). Furthermore, detrimental effects on the soil availability of K, Mg and S have been reported while the addition of PE microfibers increases soil availability of Zn (Moreno-Jiménez et al., 2022).

It is important to note that the above-mentioned effects are highly dependent on the characteristics of the plastic, including its type, concentration, size, and shape, as well as the soil type (de Souza Machado et al., 2019, Guo et al., 2024b, Zhai et al., 2024). A significant limitation in current research is the use of MP concentrations that far exceed those found in natural environments. Many studies report effects only at these artificially high levels, which are rarely encountered outside of specific pollution hotspots. At realistic environmental concentrations of MPs in soil, measurable effects may be negligible or non-existent. This insight is particularly relevant when considering potential limit values for MPs in soil.

5.1.2 Effect on soil organisms

As MNPs are widespread in the soil environment (Chapter 1), their uptake by soil organisms is inevitable, depending on their bio accessibility which is influenced by the organisms' mouth opening size. The effects of MNPs in soils is complex and multifaceted, driven by multiple factors including concentration, size, shape, and polymer type, in addition to soil characteristics (Shafea et al., 2023). Additionally, colonization by microorganisms further influences the bio accessibility and potential adverse effects on soil organisms.

Studies have confirmed the presence of MPs in earthworms, nematodes, arthropods and mollusks (Kokalj et al., 2018, Khamboonruang et al., 2024, Ju et al., 2023, Schöpfer et al., 2020, Kim and An, 2019). However, data remains scarce, with only 9 soil-dwelling species studied in risk assessments to date (Redondo-Hasselerharm et al., 2024). Earthworms are still the most studied organisms, for which PE and PVC are used the most in effect studies. This is a clear underestimation of the potential risks posed towards the soil biota from MPs. In general, effects vary depending on the studied organism, but reduced growth, survival, and reproduction, have all been demonstrated in laboratory experiments.

Earthworms, which play a vital role in soil health, have been a major focus of MNP's pollution research (Joos and De Tender, 2022). Earthworms contribute significantly to soil health by breaking down and redistributing organic matter, increasing the surface area available for microbial colonization, and playing a role in water infiltration and soil structure modification (Joos and De Tender, 2022, Lavelle et al., 2006). Earthworms can ingest and accumulate MNP particles in their digestive tract, potentially affecting their survival and reproduction. Reduced growth rates and increased mortality have been reported for *Lumbricus terrestris* when exposed to PE at various concentrations ranging from 7 to 60% dry weight (Huerta Lwanga et al., 2016). The ingestion and accumulation of MNP particles in earthworms provides an entry point for MNP into the soil food web (Huerta Lwanga et al., 2017b). Furthermore, earthworms have been shown to transport polymer particles throughout the soil matrix (Rillig et al., 2017a). In addition to earthworms, the negative effects of PS and PE, respectively, on the survival of **nematodes** such as *Caenorhabditis elegans* and **arthropods** such as springtails have also been reported (Ju, Zhu, and Qiao 2019; Lei et al. 2018). An interesting study on **snails** (*Cantareus aspersus*) examined microplastic uptake from lettuce grown under microplastic-rich conditions, finding no MPs in the snails' digestive gland but detecting them in the feces (Zantis et al., 2024).

Microorganisms are exposed to MPs in diverse ways. On the one hand, MNPs can function as physical habitats, thereby **creating selective niches** for specific microorganisms (Yu et al., 2022, Zhai et al., 2024). Microorganisms such as bacteria, fungi, algae and protozoa can form a distinct biofilm community on plastic, called the plastisphere. This biofilm formation can alter the microbial community and its function, consequently affecting its ecological role (Miao et al., 2019, Rosato et al., 2022). Remarkably, certain microorganisms have the ability to utilize MNP particles as an external carbon source for microbial assimilation, potentially enhancing cell growth and positively impacting microbial activity (Thakur et al., 2023). Conversely, the presence of MNP in the soil can exert a toxic effect on the microbial community, manifesting as both a physical and a chemical effects (Zhai et al., 2024). For example, the presence of MNP can physically impede the motility of motile microorganisms, thereby disrupting their normal environmental interactions (Cheng et al., 2023). Additionally, the presence of MNPs can cause physical damage to the cell membrane, which can in turn result in leakage of cellular content (Wang et al., 2022). Furthermore, the chemicals added to the plastic during the manufacturing process, can leach into the environment and lead to chemical toxicity (Costa et al., 2023). Indeed, plastic leachates have been demonstrated to alter microbial growth, diversity, enzymatic activity, and the abundance of pathogenic bacteria (Fei et al., 2020, Liu et al., 2022c, Li et al., 2022e).

5.1.3 Effects on plant and crops

The impact of MNPs on plant development and crop productivity is a multifaceted and variable phenomenon, as evidenced by the outcomes of current studies. The effects are contingent on both the plant species and the characteristics of the plastic used in the study (En-Nejmy et al., 2024, Okeke et al., 2023, Zhai et al., 2024, Li et al., 2023a) (Figure 16). Both positive and negative effects of the presence of MPs on plant development have been reported. The impact of MNP pollution can be categorized into direct and indirect effects (Jia et al. 2023; Okeke et al. 2023; Zhai et al. 2024).

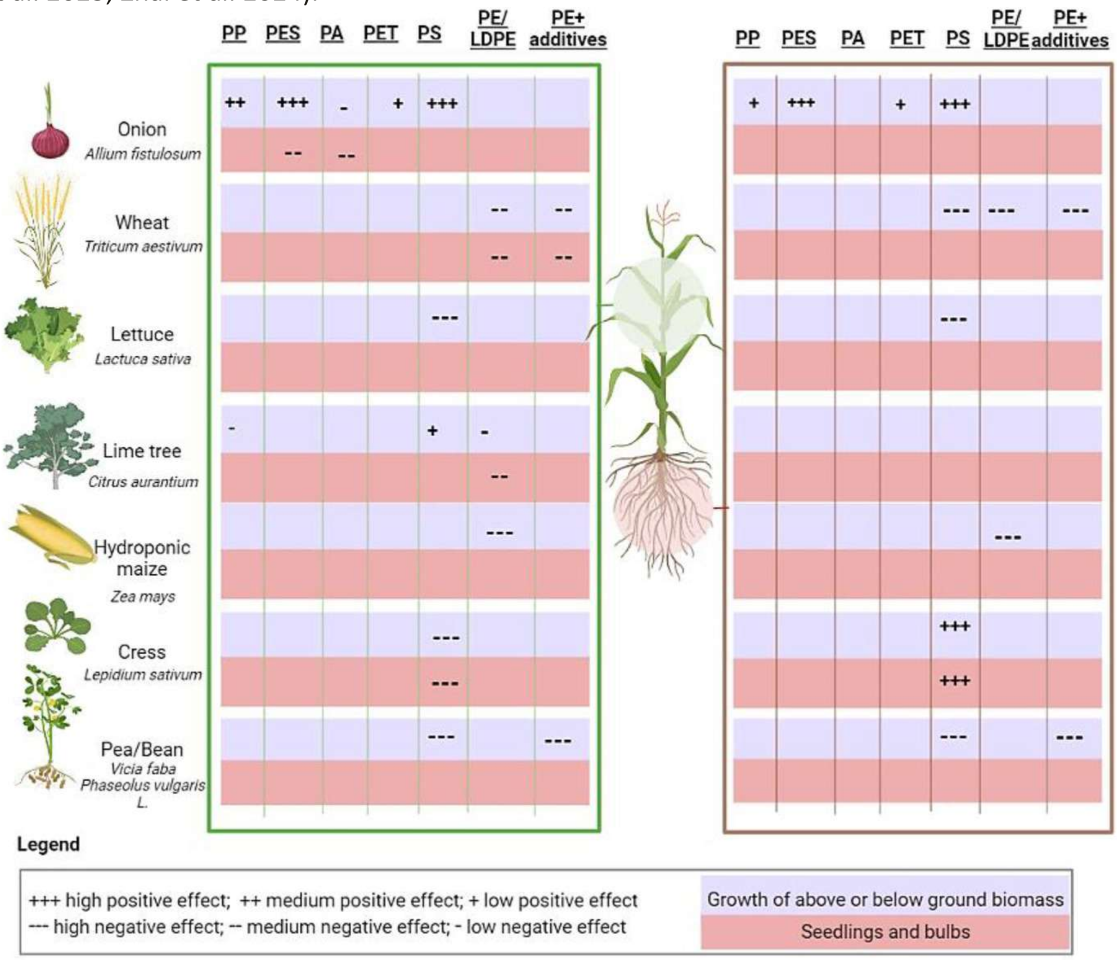

Figure 16 | Microplastics effects on various parts of plants (above ground biomass and seedlings/bulbs). Figure from Shafea et al. (2023)

Direct phytotoxic effects of MNP on plants involve the uptake, accumulation, and translocation of MNP particles within the plant, leading to various stress responses (Jia et al. 2023; Okeke et al. 2023; Zhai et al. 2024). MNP particles can enter plants via their root systems and through the stomata of the leaves via MPs in the air, subsequently moving to higher parts such as stems, fruits, and seeds where they accumulate (Li et al., 2020b, Zhai et al., 2024, Li et al., 2023a). For instance, micro-sized PS microbeads have been observed in the leaf vasculature of lettuce and wheat using scanning electron microscopy (Li et al., 2020d, Xu et al., 2022).

The size of MNPs is a critical factor in determining their impact on plants, with smaller particles having the capacity to more easily penetrate and be transported through plant tissues, resulting in greater toxicity (Li et al., 2023a). In general, a cut-off value of 5 µm particles is applied for plant accessibility as demonstrated for *Lepidium sativum* (Bosker et al., 2019) and *Hordeum vulgare* (Li et al., 2021). Furthermore, Li et al. (2020c) demonstrated that nanoplastics smaller than 100 nm could be readily taken up by plant roots and translocated to aerial parts, causing more severe physiological disturbances compared to larger MPs. However, it is noteworthy that MP particles can facilitate the entry of larger particles through the formation of deformations and distortion of the cell wall (Dong et al., 2021).

MNPs interfere with various biochemical plant processes including photosynthesis, biomass accumulation, hormone balance, nutrient metabolism, and nutrient uptake across multiple species (Pignattelli et al., 2020). The effects on plant growth and biomass are also significant. Gao et al. (2021) observed a reduction in both below- and above-ground biomass in lettuce exposed to PE fragments at concentrations ranging from 10 to 100 mg/L. Similarly, tomatoes exposed to PE and PS at concentrations ranging from 10 to 1 000 mg/L showed a decrease in root fresh weight and root length, respectively (Shi et al., 2022b). However, the responses varied considerably between studies. For example, PS MPs were found to have a negative effect on the germination percentage of cress (Bosker et al., 2019), whereas no significant effects were observed with the addition of PS to wheat or maize (Gong et al., 2021).

Indirect effects of MNPs on plants are mostly attributed to changes in soil physicochemical and microbial communities (Jia et al. 2023; Okeke et al. 2023; Zhai et al. 2024). Plants have long-standing relationships with various microorganisms, including nitrogen-fixing bacteria, such as rhizobia, and mycorrhizal fungi. These relationships often enhance plant resistance to a variety of stresses and increase nutrient uptake (Zipfel and Oldroyd, 2017). By altering the physicochemical properties of the soil, MNPs can significantly interfere with the establishment and functioning of these relationships, thereby disrupting the plant's microbial community (Zhai et al. 2024). Indeed, interference with the colonization of both rhizobia and arbuscular mycorrhizal fungi by MNPs has been reported. A study performed by He et al. (2024) reported a decrease in arbuscular mycorrhizal colonization after treatment with PET or LDPE at a concentration of 0.1% or 1% (w:w). Similarly, Wu et al. (2024) reported a significant reduction in the number of peanut nodules after treatment with PVC and PBAT at a concentration of 3% or 5% (w:w). In addition to disrupting soil properties and soil organisms, MNP particles have been shown to adhere to root surfaces, thereby interfering with water and nutrient uptake (Taylor et al., 2020).

5.2 WHAT ARE THE POTENTIAL RISKS OF MICRO- AND NANOPLASTICS ON HUMAN HEALTH?

The policy informing brief by Vercauteren et al. (2023a) provides a comprehensive overview of the current state of knowledge regarding the link between MPs, the environment and human health. This report provides a synthesis of its main points, with further details available in the original report.
The two major human exposure pathways for MPs are inhalation and ingestion. Ingestion is suggested to be the primary exposure pathway as it is linked to numerous sources, the inherent uptake mechanisms in the intestines, and the large total surface area of the digestive system (WHO, 2022). However, inhalation is increasingly recognized as a significant concern, given the high volume of inhaled air, poor ventilation in environments with abundant polymers, the large alveolar surface area, and the thin tissue barrier in the lungs (WHO, 2022). Two other exposure routes are discussed in Vercauteren et al. (2023a), with dermal contact being a very limited exposure pathway due to the physical barrier of the skin and the hydrophobicity of plastics. Only smaller particles (< 100 nm) can cross the skin barrier in healthy skin (Bouwstra et al., 2001). A new suggested exposure route is via infusion in the medical sector, by the usage of plastic products used for infusion therapy (tubes, IV-bags, syringes) (Gopinath et al., 2022).

Regarding ingestion, MNPs have been detected throughout the food chain, including commercial **fish**, **bottled water**, beer, honey, and tea (**Table 13**). Limited studies have also shown MPs in agricultural related products such **as fruits, vegetables**, grains, cereals, spices and terrestrial animal products (**Table 13**).

Table 13 | Distribution levels and size of micro- and nanoplastics for fruits, vegetables and food. *IQR* interquartile range.

Species	Common name	Micro- and nano-plastics Mean ± SD	Size (µm) Median (IQR)	Reference
Malus domestica	Apple	195 500 ± 128 687 MP/g	2.17 (1.56–3.19)	(Conti et al., 2020)
Pyrus communis	Pear	189 550 ± 105 558 MP/g	1.99 (1.87–2.59)	(Conti et al., 2020)
***Brassica oleracea* var. italica**	Broccoli	126 150 ± 80 715 MP/g	2.10 (1.86–2.95)	(Conti et al., 2020)
Lactuca sativa	Lettuce	50 550 ± 25 011 MP/g	2.52 (2.18–2.78)	(Conti et al., 2020)
Daucus carota	Carrot	101 950 ± 44 368 MP/g	1.51 (1.36–2.00)	(Conti et al., 2020)
Sardinia pilchardus	Sardines	4.63 MPs per individual	5.27-1 310	(Renzi et al., 2018)
	Bottled water	10.4 MP/L	> 100	(Mason et al., 2018)
	Bottled water	325 MP/L	6.5 – 100	(Mason et al., 2018)
	Beer	1 212-9 659 MPs/100 mL		(Li et al., 2022d)
	Honey	32-108 fibers/kg	1-30	(Mühlschlegel et al., 2017)
	Tea	11.6 billion MP/cup	0.01 – 150	(Hernandez et al., 2019)
	Milk	6.5 ± 2.3 MP/L	100 – 5 000	(Kutralam-Muniasamy et al., 2020)
	Table salt	50-280 MPs/kg	10-3 500	(Iñiguez et al., 2017)

MNPs have been detected in a wide range of human tissues, including feces, placenta, lung tissue and blood (Roberts et al., 2022, Niu et al., 2023, Ragusa et al., 2021, Schwabl et al., 2019) **(Figure 17**). Evidence suggests that MNPs are distributed throughout the body, and can result in their accumulation in organs and tissues (Pitt et al., 2018). Elimination occurs through feces, urine and exhalation, however efficiency varies according to particle characteristics and the individual.

Studies on the effects of MNPs on human health are limited due to limitations in human tissue sampling and detection methods (Feng et al., 2023). Laboratory experiments using human cell lines, tissues and animal models have shown that exposure to MNPs can induce inflammatory responses, metabolic disorders and affect both gastrointestinal and liver health (Khan and Jia, 2023, Niu et al., 2023, Li et al., 2022c). However, often these experiments use relatively high concentrations of particles that may not sufficiently resemble the abundance and types of particles that humans are exposed to (Gouin et al., 2022). These in vivo experiments are also short-term, while the impact on humans is likely to be a long-term chronic exposure (WHO, 2022).

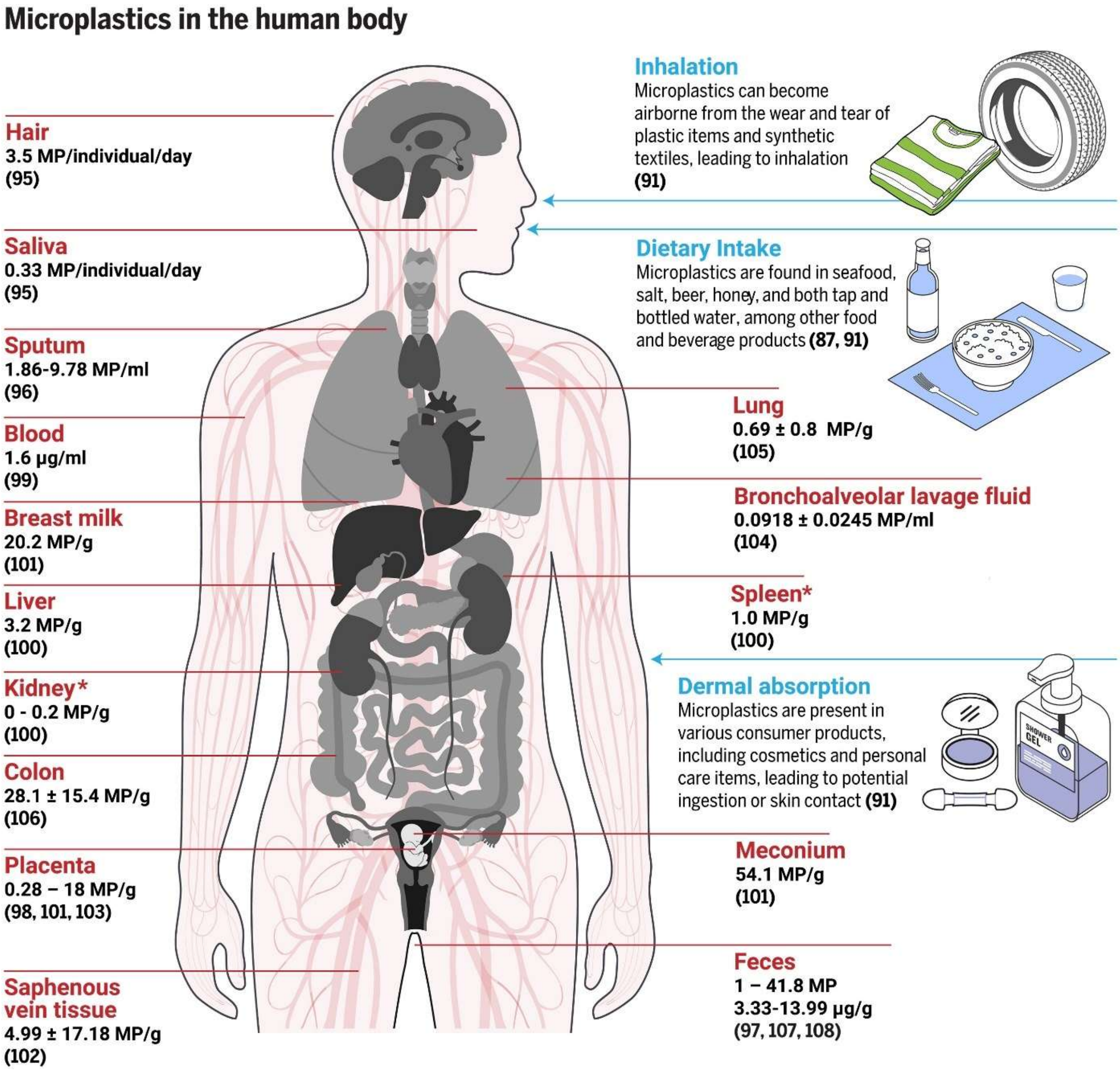

Figure 17 | Locations in the human body where microplastics have been reported. Exposure pathways (turquoise labels) and reported quantities (red labels) are shown. Quantities of microplastics are as reported in each study and have not been further quality assurance and quality control–screened for this review. Intercomparisons should be made with caution because of variation in methods and units of reporting between studies. Because some methods do not characterize individual particles, it is likely that quantities reported by mass relate to both micro- and/or nanoparticles (see section Methodological advances for discussion). *Quantities reported as being around the limit of detection. Source (Thompson et al., 2024).

5.3 WHAT ARE THE LIMIT VALUES FOR MICROPLASTICS ABOVE WHICH ADVERSE EFFECTS HAVE BEEN MEASURED?

Understanding the impacts of MPs on the environment and humans is increasingly critical, highlighting the need to assess their risks to soil organisms and humans under realistic exposure conditions. Unfortunately, harmonized guidelines or protocols to properly validate MNP (eco)toxicity testing are not yet available. Due to the microplastic complexity and heterogeneity, there are significant challenges in effective testing and assessment methods coupled. Variations in chemical composition, aging processes, and environmental weathering further complicate these efforts (Thompson et al., 2024). **Early laboratory studies (and even current studies) often focus on high concentrations with monodisperse plastics, not representing environmental samples.** While these studies can give a mechanistic insight, this discrepancy between experimental designs and environmental exposures, including the overrepresentation of specific polymers and species, have underscored the necessity of testing at environmentally relevant concentrations.

Efforts are made to provide a framework to improve risk assessments for MPs. An in-depth overview of all components regarding risk assessment is given in Koelmans et al. (2023). Four critical areas are emphasized: (1) physical particle characterization, including the entire microplastic continuum (Koelmans et al., 2022), (2) chemical properties of MPs, (3) defining and calculating ecological risks for micropolymer particles and (4) integration with existing scientific and policy frameworks.

Significant progress has been made in aquatic systems, where risk assessments have utilized extensive databases and a diverse array of scientific studies. Although soil research on MPs lags behind, Redondo-Hasselerharm et al. (2024) has developed a first risk assessments for soils based on data of 51 studies. This study emphasizes the importance of considering variations in MP characteristics among various sources, such as mulching, compost, sewage, and background pollution. By applying strict QA/QC screening tools (as described in Chapter 4.3) and data alignment methods, they addressed the challenge of non-alignment between existing exposure and effect data.

Briefly, this risk assessment is based on two factors: how much microplastics are in the environment (exposure risk) and how sensitive different species are to them (effect risk). To measure the potential harm, researchers use thresholds like the "no observed effect concentration" (NOEC), which shows the highest microplastic concentration that doesn't cause harm, and the "lowest observed effect concentration" (LOEC), which indicates the lowest level that does cause harm. From this, a "hazardous concentration" (HC5) is calculated, showing the level where 5% of species may be affected. The safety threshold, called the "predicted no effect concentration" (PNEC), is then set. If the environmental concentration exceeds this threshold (RCR > 1), it signals potential ecological harm. This method helps to measure and compare the risks microplastics pose to different ecosystems like marine, freshwater, and land environments.

Risk assessments for MPs combine exposure and effects to assess their potential harm to human, plant, and ecosystem health.

Exposure risks are based on environmental microplastic concentrations, while for effect on ecosystem health species sensitivity is considered. Species sensitivity distributions (SSDs) are used to estimate species responses to microplastic exposure, with the hazardous concentration for 5% of species (HC5) being a key metric. SSD-based risk assessments for MPs in various environmental compartments have been conducted, revealing different levels of concern across marine, freshwater, and terrestrial ecosystems.

- In the **marine environment**, Everaert et al. (2018) calculated a safe concentration of 6 650 floating MP particles/m^3 based on SSD using data based on 14 marine species. While current average concentrations of floating MPs are generally below this threshold (0.2 - 0.9 particles/m^3), some heavily polluted areas, particularly coastal regions and narrow straits, are already approaching or exceeding this level (e.g. for the NE Pacific seawater concentrations of 8 - 9 200 particles/m^3 were found). This indicates localized risks in these high-concentration zones.
- **Beach environments** are particularly vulnerable to microplastic accumulation. A safe concentration is estimated to be 540 particles/ kg sediment (Everaert et al., 2018). From this study, it is estimated that between 32 and 144 particles/ kg dry sediment are found on the deposition zone of Belgium beaches. This accumulation is expected to increase over time, potentially leading to higher risks in the future.
- For **deep sea sediments**, the same safe concentration as for beach sediment (540 particles/ kg sediment) is used (Everaert et al., 2018). Estimated current concentrations are estimated to be 1.5 - 6.7 particles/ kg sediment.
- In **freshwater sediments**, Redondo-Hasselerharm et al. (2023) derived SSDs for 14 freshwater benthic species. Safe concentrations were calculated up to 4.9×10^9 ($6.6 \times 10^7 – 1.9 \times 10^{11}$) and 1.1×10^{10} ($3.2 \times 10^8 – 4.0 \times 10^{11}$) particles/kg dry sediment. While on average, freshwater sediments have concentrations below these thresholds, some studies have reported values approaching the lower confidence intervals. For example, a river in China recorded concentrations of 1.5×10^5 particles/kg dry sediment (Xia et al., 2021).
- Redondo-Hasselerharm et al. (2024) conducted the first risk assessment for MPs in **soils**, albeit with a limited number of studied organisms (9). They reported safe concentrations ranging between 4.0×10^7 and 2.3×10^8 particles (1-5000 µm)/kg of dry soil for different MP sources. This study highlighted the importance of differentiating between MP source materials (e.g., compost, sewage sludge) when assessing risks, as each source may present unique hazards and exposure patterns. In Chapter 3, we described that values are currently ranging between 0.34 up to 12×10^6 MPs/kg soil.

So far, no risk assessments for **human health** have been developed (Vogel et al., 2024). However, risk assessment frameworks for human exposure are being developed within the EU funded project POLYRISK (polyrisk.science).

5.4 ARE THERE CERTAIN ADDITIVES OR TYPES OF PLASTIC THAT CLEARLY HAVE A HIGHER RISK TO CAUSE ADVERSE EFFECTS?

Within the Plast-Chem project, 15 chemical groups were identified as of high concern. Below is a list of additives described that are present in consumer goods and have a known impact on human health (Interreg Baltic Sea Region, 2024).

- **Bisphenols**: Most used chemical within this group is bisphenol A (BPA), present in 3 Mt produced per year such as the lining of aluminum cans. These are often found in PC, resins, PP, PE and PVC. BPA is suspected to disrupt hormone balances, but is also linked to testicular cancer, obesity and reproductive disorders (Vom Saal and Vandenberg, 2021).

- **Phthalates:** Are mainly present in PVC to add fragrance to products and make them more pliable. Some of those phthalates are associated with endocrine disruptors, breast cancer, developmental issues, decreased fertility, obesity and asthma. Phthalates can easily leach into the environment (Wang et al., 2021b, Henkel et al., 2022).
- **Flame retardants**: Used in electronics, insulation material but also (transportation) furnishing. They are toxic and do not break down easily in nature. They are linked with endocrine and thyroid disruption, immunotoxicity, reproductive toxicity, cancer, and adverse effects on fetal child development and neurological function (Castorina et al., 2017, Doherty et al., 2019, Feiteiro et al., 2021).
- **Perfluorinated chemicals (PFCs)**: Often found in commercial and household products such as clothing, textiles, non-stick cooking surfaces, fast food and microwavable food packaging. They are linked to liver, kidney, brain and spleen. In animal studies, PFCs cause cancer, neonatal mortality, delays in physical development, and endocrine disruption. Higher maternal levels of PFCs are associated with delayed pregnancy, while in the case of men, higher PFC levels could compromise the integrity of the reproductive system (Carnero et al., 2021).

6 CAN MICROPLASTICS AMPLIFY THE EFFECT OF CHEMICALS THAT MAY ADHERE TO THEM?

MNPs influence their surroundings directly when they enter soil environments, but their role doesn't stop there. They can also interact with and significantly affect the behavior of other contaminants. Due to their high specific surface area and hydrophobic nature, MPs are highly effective at adsorbing pollutants. This means they can serve as a vector, carrying these pollutants through various ecosystems (Peña et al., 2023). This is concerning as other immobile contaminants, which normally interact strongly with the soil matrix, can become mobile and can be carried with the pore water flow to deeper soil layers or groundwater (Hale et al., 2020).

Environmental significant transport of contaminants on MNPs can apply under four conditions: (i) polymer particles must be of sufficiently high concentrations, (ii) particles must be more mobile than the (non-sorbed) contaminant, (iii) the contaminant must be of concern, and (iv) the desorption of the contaminant during the travel time of the MNP must be low (Castan et al., 2021). The mobility of MNP particles in soil depends on flow conditions, solution chemistry, and physicochemical properties of the soil and the particle (Gao et al., 2006). To reach maximum MNP transport, the attachment efficiency towards the soil matrix must be zero, which might be found for short travel and time scales. The significance of MNP-facilitated contaminant transport is ultimately determined by the rate of contaminant desorption when MNPs are highly concentrated, mobile, and carrying contaminants of environmental concern. This process is driven by differences in the substance's tendency to move or transfer from one phase (plastic) to another (soil environment).

Castan (2021) found that under slow soil water flow (<1 m/year), organic contaminants desorb from small particles before reaching deeper soil layers, limiting the relevance of MNP-facilitated transport. In contrast, fast flow conditions after droughts followed by heavy rain create larger transport channels, allowing MNPs to move rapidly with contaminants. However, the process depends on soil properties, with light-textured soils and macropore-dominated transport enabling faster contaminant movement, while heavy-textured soils and matrix-dominated transport retain contaminants more effectively. Castan et al. (2021) concluded that MNPs generally do not enhance contaminant mobility in farmland soils, as desorption is too quick to be environmentally relevant, even in fast flow conditions.

7 BEHAVIOUR OF MICROPLASTICS IN SOILS AND SEDIMENTS

7.1 HOW DO MICROPLASTICS BEHAVE IN SOILS AND SEDIMENTS?

One of the primary soil properties that determines the spatial distribution and dynamics of MP is the soil structure (Guo et al., 2022). Soil aggregation constitutes the initial stage in the soil formation process and results from the rearrangement of individual particles in combination with processes like flocculation and cementation (Duiker et al., 2003, Bronick and Lal, 2005, Payne, 1988). Soil aggregates are the building blocks of the hierarchical organization of soil structure, with macro-aggregates (> 250 µm) comprising smaller micro-aggregates (20-250 µm), that in turn consist of primary soil particles (Tisdall and Oades, 1982). Both clay and soil organic matter (SOM) play a crucial part in the soil aggregation process. Soil aggregates in turn disintegrate due to mechanical stresses because of drying-and-wetting cycles, frost or anthropogenic influences such as ploughing and compaction by heavy machinery (Rücknagel et al., 2012, Six et al., 2004). Macro-aggregates are more susceptible to disintegration because of the more transient nature of the organic binding agents (fine roots and hyphae) as compared to the microbial debris that hold together the microaggregates. Micro-aggregates are more persistent and can exist both within macro-aggregates and independently in the soil matrix.

Soil aggregation protects particulate organic matter (POM), which consists of relatively large organic particles, from biodegradation through physical protection and the reduction of O_2 diffusion in stable aggregates (Bertini and Azevedo, 2022). A similar assumption can be made for MP, as they typically contain around 80% organic carbon (Rillig et al., 2021) and can thus be considered as a specific, recalcitrant type of POM. As micro-aggregates are more stable than macro-aggregates, POM and thus MP that reside within micro-aggregates are less bio-available than that in macro-aggregates. Incorporation of MP in soil aggregates may thus also limit microbial degradation and protect them from uptake by soil fauna (Hurley and Nizzetto, 2018, Rillig et al., 2017a) and will also influence MP mobility throughout the soil profile (Mueller et al., 2012). However, soil aggregation is a highly dynamic process (Totsche et al., 2017) and the distribution of MP over the aggregate fractions may vary strongly over time. As such, MP can be stored inside soil aggregates and subsequently re-released when these start to disintegrate.

Field surveys and monitoring campaigns have pointed out that MP can get incorporated within the soil aggregates in varying proportions according to shape and polymer type. Zhang and Liu (2018) observed that 72% of polymer particles (of which 95% were MP) in agricultural soils (Dian Lake, SW China) were associated with soil aggregates, while only 28% were present in the inter-aggregate pore space. Large macro-aggregates held most of the MP (34%), while only smaller proportions were present in the smaller macro-aggregates and micro-aggregates (22 and 16%, respectively). Films and fragments were more frequently found inside large macro-aggregates, while fibers were more abundant in the inter-aggregate space. Liu et al. (2023b) observed a predominance of film and granule-shaped MP in macro-aggregates on mulched agricultural fields in the Hubai Province of China, while fibers were more abundantly found inside the micro-aggregates.

At the same time, the presence of the MP themselves influences soil structure. A small number of controlled experiments have been undertaken to study the effects of MP addition on soil aggregation processes. Lehmann et al. (2019) studied the effects of polyester fibers on soil aggregation and water-stability of the aggregates in combination with wetting-and-drying cycles and soil biota (size = 5 mm, 0.1 wt%), finding no effect of the MP on the formation of the aggregates itself, but a reduction in aggregate stability when soil biota were added in combination with MP. Other studies found a reduction of water-stable aggregates (de Souza Machado et al., 2018a, Liang et al., 2021) and reported decreasing soil organic carbon concentrations in macro-aggregates upon MP addition (Zhang and Zhang, 2020). However, according to de Souza Machado et al. (2018b) MP fibers can effectively establish a tighter connection between microaggregates because of their linear shape, while MP fragments interact more loosely with the soil. Nonetheless, still very few controlled experiments or field surveys have studied the spatial distribution over different aggregate size fractions and the evolution of this process over time (Zhang and Liu, 2018, Zhang et al., 2019, Zhang and Zhang, 2020). Controlled experiments have focused on larger MP fragments (> 2 µm) and no studies have examined how small MP of different sizes would distribute over different aggregate size fractions yet.

7.2 WHAT DISTRIBUTION PATTERNS ARE KNOWN AND HOW FAST DO THEY MIGRATE THROUGH SOILS?

Concentrations of MP stored in soil decrease with soil depth (Liu et al., 2018). Samples taken at agricultural sites in northern Germany show a gradual decrease in concentration with depth, with three times as many MP concentrated in the upper 10 cm of the soil profile compared to the layer between 20 and 30 cm (Harms et al., 2021). Nevertheless, MP migrate downwards through the soil profile, which could eventually lead them to reaching the groundwater table where they risk contaminating drinking water supplies (Ren et al., 2021, Ya et al., 2021, Viaroli et al., 2022). However, the downward movement of MP towards the groundwater table has never been proven in field experiments (Re, 2019). Nonetheless, their presence has already been attested in the groundwater. MP from eight types of polymers were discovered in an aquifer in Australia with an average concentration of 38 items/L (Samandra et al., 2022). However, according to Viaroli et al. (2022), MP in groundwater may originate not only from soils but also from atmospheric deposition through contact with surface waters that replenish aquifers or infiltration systems and urban recharge (domestic and industrial sewers and urban surface runoff). The potential of MP reaching the groundwater table through the soil largely depends on the thickness of the unsaturated zone, which acts as a buffer (Keller et al., 2020, Scheurer and Bigalke, 2018). Soil porosity is of primary importance in this process, which is in turn determined by soil texture. Soil texture is a determining factor for other soil properties like soil structure, pH and soil biota, which also affect MP mobility (Boots et al., 2019). The mobility of MP is not only influenced by soil physical and chemical properties, but also by the properties of the MP themselves, like size and shape. According to Yu and Flury (2021), logically, the smallest fraction (< 10 µm) has the greatest potential to be transported down through the soil. Shape also plays a significant role. According to (Hu et al., 2022), fibers are equally distributed over different depths, but films are less abundant in deeper soil layers. On the other hand, fibers are less susceptible to downward transport than spherical or granular MP because they can change their orientation as they move and as such are blocked more often (Keller et al., 2020). They also show a tendency to form aggregates with soil particles due to their linear properties, thus enhancing the formation of macropores (Zhang et al., 2019). Biological agents can influence and accelerate the migration process. An important agent for the transport and movement of MP in the soil is bioturbation by plants and soil organisms. Earthworms were proven to play an important role in the downward transport of MP in the soil (Rillig, 2012, Huerta Lwanga et al., 2017a). Bioturbation experiments in microcosms have proven that ingestion and subsurface excretion by *Lumbricus terrestris* are the main drivers for the vertical transport of nanoplastics (NP, < 1µm) in the soil (Heinze et al., 2021). The NP are specifically concentrated along the burrow walls. Crop roots also exert an effect on the movement of MP, as plant roots can either retain MP or even move them upward (Li et al., 2020a). Human activities, like mouldboard ploughing or harvesting crops with an important biomass below the surface (e.g. carrots or potatoes), can also promote the downward movement of MP (Rillig et al., 2017b).

Processes of adsorption and desorption also play an important part in the transport of micropolymer particles, as well as galleries and pores that are created by plant roots (Yooeun and Youn-Joo, 2018). Adhesion of surfactants decreases the hydrophobicity of the MP surface and enhances the mobility of MP (Jiang et al., 2022). Very small micropolymer particles can also move along with water as a colloid (Rillig et al., 2017a). Little is known about the speed of the transport of polymer particles. Column experiments have proven that small MP leach out faster in sandy soils when they are exposed to subsequent wetting and drying cycles (O'Connor et al., 2019). Surface charge might significantly determine how MP behave in the soil as their retention correlates in a significant manner with the soil zeta potential (Wang et al., 2021c). PVC reaches an isoelectric point at a pH of 6.43, while also depending on humidity. PET MP on the other hand exhibits very variable zeta potentials across a pH of 2.5-7.0. However, surface chemistry changes upon MP weathering, which makes such theoretical considerations even more uncertain. And then we also have the MP corona which may completely alter the behavior of MP in soils

8 REGULATIONS

8.1 DO REGULATIONS REGARDING PLASTIC POLLUTION ALREADY EXIST FOR OTHER ENVIRONMENTAL COMPARTMENTS BESIDES THE SOIL COMPARTMENT?

A global regulatory framework for MPs in soil, groundwater and sediments remains limited, lagging efforts focused on aquatic ecosystems which have been set in place since 1970 (Munhoz et al., 2022) (Figure 18). As highlighted in the review by Munhoz et al. (2022), it took seven decades after the invention of synthetic plastic for international conventions to begin addressing plastic pollution. Initially, most regulations targeted plastic pollution broadly, but from the mid-2000s to 2020, they increasingly focused on MPs. Figure 18 illustrates the timeline of policies addressing MPs contamination, along with key regulations specifically targeting MPs.

While significant strides have been made to address marine plastic pollution through these regulations, there are also measures being implemented that target broader environmental concerns related to plastic pollution. For example, bans on microbeads in personal care products such as the US Microbead-Free water act (2015)

(McDevitt et al., 2017) along with similar regulations in Australia (Plastic Reduction and Circular Economy Act 2021), New Zealand (Environmental Protection Authority, 2018) and the EU have been implemented (European Commission, 2021). China announced the Prevention and Control of Waste Plastic Processing and Utilization initiative, aiming to regulate plastic waste management and reduce environmental pollution caused by plastic processing and reuse (Zhang and Liu, 2018). The prohibition of certain single-use plastics in the EU have gained global attention (McDevitt et al., 2017, Saini and Sharma, 2022). The EU has also taken significant steps with the Zero Pollution Action Plan, which includes measures to reduce microplastic pollution by 30% by 2030. This includes restrictions on MPs intentionally added to products, and although this primarily impacts urban wastewater systems, they also have implications for soil and groundwater, as MPs can migrate from treated wastewater and sludge applied to agricultural lands. Therefore, the plan's comprehensive approach contributes to reducing microplastic pollution across various environmental compartments (European Commission, 2021). These regulations aim to mitigate the overall environmental impact of plastics by reducing the entry of harmful materials not only into aquatic ecosystems, but also into terrestrial and atmospheric environments. As more nations adopt these measures, they represent a growing acknowledgment of the pervasive effects of plastic pollution across all ecosystems—not just the oceans.

However, despite these advancements, regulatory frameworks for managing (micro)plastic pollution in terrestrial environments, such as soil and sediments, remain underdeveloped, and more comprehensive policies are needed to address these emerging threats comprehensively.

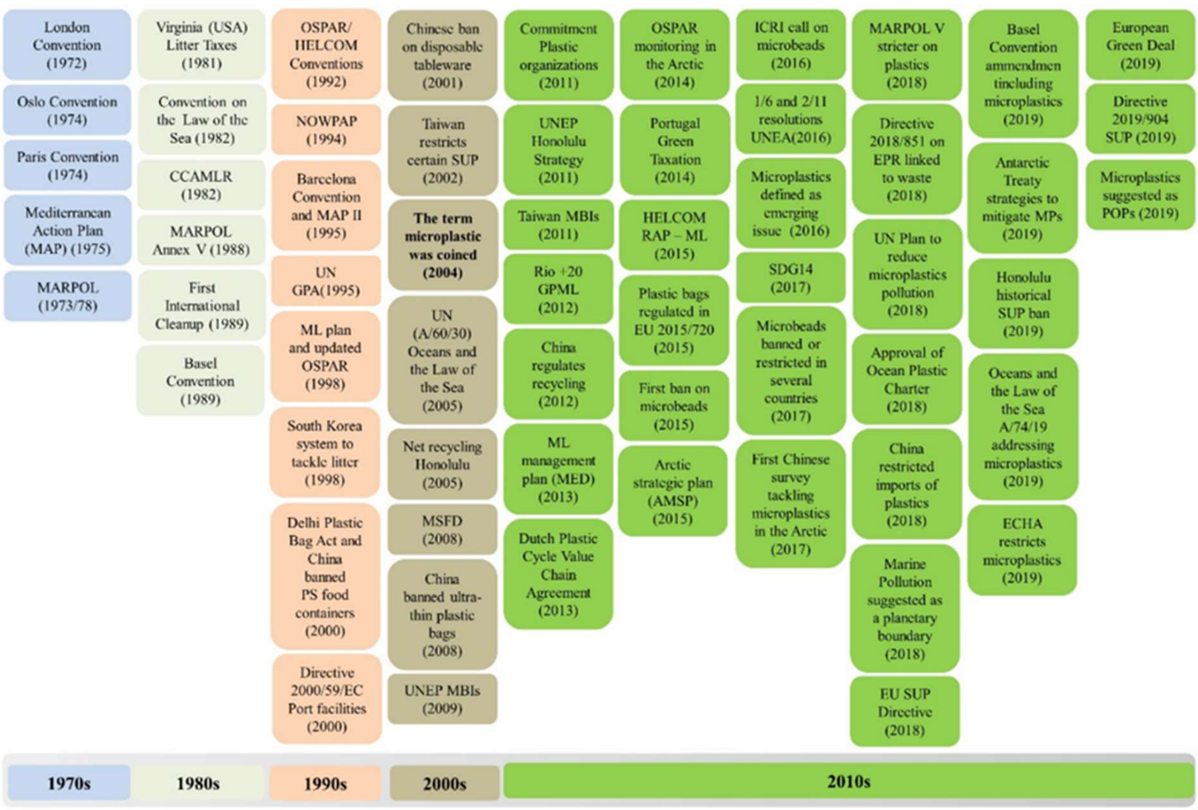

Figure 18 | Timeline of policies targeting plastic and microplastic contamination. **Source Munhoz et al. (2022).**

8.2 ARE THERE ALREADY ENVIRONMENTAL REGULATIONS ON MICROPLASTICS IN SOIL, GROUNDWATER, AND SEDIMENTS IN OTHER COUNTRIES?

Currently, environmental regulations targeting (micro)plastic pollution in soil, groundwater, and sediment are limited and vary by region. While some countries have started addressing (micro)plastic pollution in the soil environment, these regulations are often indirect or focus on broader plastic management rather than specifically targeting these environments (Table 14).

The EU stands as a pioneer in regulating MPs, with measures targeting environmental entry and potential expansion into agricultural and sedimentary contexts under the Zero Pollution Action Plan implemented in 2021 (European Commission, 2021). Efforts include reformulation on products like controlled release fertilizers and detergents (European Commission, 2023).

In January 2019, ECHA has proposed restrictions on intentionally added MPs to reduce environmental contamination, including those in controlled release fertilizers (ECHA, 2025) (Table 14). A detailed examination of these proposals and their impact is outlined in ECHA's Annex XV Restriction Report, which discusses the regulatory process, biodegradability criteria, and potential implications for various industries, including agriculture (ECHA, 2019). The implementation of these restrictions involves consultations with stakeholders to ensure the measures are practical and address industry-specific challenges. This includes examining biodegradable alternatives and testing standards for microplastic-containing products (ECHA, 2019). The EU Commission adopted the restriction on 25 September 2023. The first measures, for example the ban on loose glitter and microbeads, start applying on 17 October, when the restriction enters into force. In other cases, the sales ban will apply after a longer period to give affected stakeholders the time to develop and switch to alternatives (European Commission, 2023).

In contrast, the U.S. Microbead-Free Waters Act (2015) targets microbeads in cosmetics, indirectly reducing MPs in irrigation wastewater but lacks soil-specific regulations. Similarly, Australia and New Zealand have banned microbeads in personal care products but lack targeted laws for MPs in soils or sediments, despite recognizing risks from treated wastewater. China's focus on single-use plastic bags and recycling indirectly mitigates microplastic pollution, though not specifically in agricultural or sedimentary contexts. Countries like Japan and South Korea rely on robust waste management systems to reduce macroplastic waste, with limited focus on microplastic soil contamination.

Table 144 | Overview of international regulations and policies targeting microplastic pollution

Region	Year	Regulation	Scope	References
European Union	2021	Zero Pollution Action Plan	Reduce microplastic pollution by 30% by 2030	(European Commission, 2021)
	2019	The European Chemical Agency (ECHA)	proposed restrictions on intentionally added MPs in products.	(ECHA, 2019)
United States	2015	Microbeads-Free Water Acts	Microbeads banned in cosmetics	(Microbead-Free Waters Act, 2015)

Australia	2021	New South Wales: Plastic Reduction and Circular Economy act 2021	Bans on microbeads in cosmetics and other personal care products	(Plastic Reduction and Circular Economy Act, 2021)
New Zealand	2018	Environmental Protection Authority	Bans on microbeads manufacture and sell in cosmetics and other personal care products	(EPA, 2024)
China	2019		Implemented policies focusing on reducing plastic waste through bans on single-use plastics and promoting recycling	(Zhang and Liu, 2018)
Thailand	2019	Ministry of Public Health	Banning of the import, production, and sale of cosmetic products containing microbeads	(ChemLinked, 2019)
UNEP Honolulu	2011	UNEP Honolulu strategy	Reduce sea-based and land-based pollution	(Pettipas et al., 2016)
Netherlands	2012	Dutch plastic cycle value chain agreement	To close the loop on ship-generated waste by prevention of waste and by delivering ship-generated waste at ports prior to sailing, wherever possible	(Kamp et al., 2014)
Canada	2021	Canadian environmental protection act	Canada list of toxic substances	(Administrator of the Government of Canada in Council, 2021)

8.3 ARE THERE REGULATIONS UNDER DEVELOPMENT THAT COULD SERVE AS EXAMPLES?

Researchers working on (micro)plastic pollution in soil ecosystems are therefore urging for new legislation. On the 8th of April, there will be a science-policy briefing during the AgrifoodPlast conference held in Brussels, in which we aim to convince the European Union to include plastic pollution in the upcoming Soil Monitoring Law ((34) Bridging Science and Policy: The Crucial Decision on Plastic Pollution in Soil | LinkedIn). This would be a first step towards the regulatory frameworks needed.

The United Nations Environment Programme (UNEP) is leading efforts to formulate a legally binding global treaty aimed at addressing plastic pollution throughout the entire lifecycle of plastics, from production to disposal. Negotiations have been ongoing, with the Intergovernmental Negotiating Committee (INC) conducting multiple sessions:

- INC-1 (Uruguay, 2022): Laid the groundwork for negotiations, emphasizing the need for a comprehensive treaty.
- INC-2 (France, 2023): Discussed global obligations like bans on certain plastics and measures for microplastics.
- INC-3 (Kenya, 2023): Focused on financial mechanisms to support developing countries.
- INC-4 (Canada, 2024): Worked on monitoring and reporting systems for plastic pollution.
- INC-5 (South Korea, 2024): Aimed to finalize the treaty text with binding targets and timelines.
- Extra Session (Switzerland, 2025): Will address remaining issues to adopt the treaty by the end of 2025.

The existing and emerging regulations targeting microplastic pollution are important steps but are not enough to comprehensively tackle the issue. Most regulations focus on specific sources, such as microbeads in cosmetics, or particular environments, mainly marine systems, leaving soil, groundwater, and sediments less regulated. Additionally, the current regulatory landscape is fragmented, some countries and regions enforce strict measures while others lack even basic regulations. This inconsistency facilitates the cross-border movement of microplastics, highlighting the need for a legally binding global treaty with clear targets and consistent standards. While the Global Plastics Treaty under negotiation shows promise, it is not yet enforced. Moreover, current regulations often focus on end-of-life solutions, emphasizing waste management and cleanup rather than reducing plastic production or promoting sustainable alternatives. Scientific and monitoring challenges also pose significant barriers. Standards for detecting and measuring microplastics in soil, groundwater, and sediment are still under development. Without reliable and standardized data, setting effective regulations becomes difficult. To strengthen regulations, it is essential to adopt a comprehensive global treaty with legally binding commitments to reduce plastic production and manage waste across all environments, including soil. A lifecycle approach to regulations, addressing all stages from production and use to disposal and cleanup, is necessary. Additionally, providing financial support and technology transfer to developing countries and establishing unified science-based standards for monitoring microplastics would significantly enhance regulatory effectiveness. In conclusion, while current regulations are valuable, a more comprehensive, globally coordinated, and lifecycle-based approach is essential to effectively tackle microplastic pollution.

8.4 ARE THERE ALREADY EXAMPLES OF MONITORING REQUIREMENTS FOR MICROPLASTICS IN ENVIRONMENTAL COMPARTMENTS?

Monitoring frameworks for MPs differ significantly in scope, enforcement, and standardization across countries and environmental compartments. This section distinguishes between **legally binding monitoring requirements** and **project-based or voluntary national initiatives**, highlighting the current state of microplastic monitoring globally.

8.4.1 Legally binding monitoring requirements

Monitoring requirements established through legislation or binding directives play a crucial role in systematically tracking MPs and informing policy.

The **Marine Strategy Framework Directive** (MSFD) – EU requires member states to monitor and report on MPs as part of Descriptor 10 (marine litter) (Galgani et al., 2013). This includes tracking polymer particles in seawater, sediments, and biota. The new EU MSFD Guidance on Monitoring Marine Litter in the European Seas provides protocols for monitoring of marine litter based on research developments and Member States efforts to increase the comparability of data and assessments (Galgani et al., 2013). For Belgium specifically, this is conducted by VLIZ (seawater and sediments) and ILVO (biota).

The **United Nations Environment Program** (UNEP), in collaboration with UNESCO's IOC, provides guidelines for monitoring plastic litter in marine environments. These guidelines standardize sampling and analysis of plastic debris, including MPs, in water, sediment, and biota. The aim is to harmonize methodologies, enabling international comparisons and supporting global efforts to address marine plastic pollution (Cheshire et al., 2009).

The **National Oceanic and Atmospheric Administration (NOAA)**, USA has developed standardized protocols for monitoring MPs in marine environments. These protocols encompass surface water sampling, sediment analysis, and the evaluation of plastics within the food chain. The methodologies are detailed in the "Laboratory Methods for the Analysis of Microplastics in the Marine Environment," which provides recommendations for quantifying synthetic particles in waters and sediments (Masura et al., 2015).

The "**Guidelines for Harmonizing Ocean Surface Microplastic Monitoring Methods**" were developed in Japan to standardize methodologies for monitoring MPs at the ocean surface, ensuring comparability of results across different studies and regions. These guidelines provide detailed protocols for sample collection, handling, processing, and analysis, as well as reporting requirements. The primary goal is to facilitate a unified approach to monitoring microplastic densities, thereby enhancing the quality and comparability of data used to assess the extent of marine plastic pollution. The guidelines were first published in May 2019 and have undergone revisions to improve their effectiveness. The latest version, released in November 2023, includes updates to definitions and categories of fundamental data items, aiming to further harmonize monitoring efforts and data sharing. These guidelines are a collaborative effort, with contributions from international experts and organizations, and are intended to support global initiatives in combating marine plastic pollution. They are available for download on the Ministry of the Environment, Japan's website (Yutaka et al., 2023).

8.4.2 National monitoring initiatives

The **MICROSOF project** conducted the first national assessment of microplastic contamination in French soils (Palazot et al., 2024). Researchers analysed 33 soil samples from the French soil quality monitoring network (Palazot et al., 2024), predominantly from agricultural areas, to determine microplastic presence and characteristics. This study found that 76% of the analysed soil samples contained MPs, with concentrations ranging from less than 6.7 to 80 particles/ kg dry soil. Contamination was widespread in agricultural areas such as croplands, grasslands, vineyards, and orchards, while only one forest sample showed microplastic presence, suggesting higher contamination in soils exposed to agricultural practices. The abundance of MPs in French soils was consistent with levels found in similar studies, indicating an intermediate level of contamination. The study highlights the need for further monitoring to understand the sources and environmental behaviour of MPs in soils. Integrating microplastic analysis into national soil monitoring programs is recommended to assess potential risks to ecosystems and human health (Palazot et al., 2024).

Environment and Climate Change Canada (ECCC) has been actively involved in monitoring MPs in both freshwater and marine systems, with particular attention to the Great Lakes and Arctic waters. In the Great Lakes region, studies have documented the presence of MPs in water, sediment, and wildlife, highlighting the need for standardized monitoring strategies to assess contamination levels and inform management actions. These initiatives employ standardized sampling and analysis techniques to ensure the collection of reliable data, which is crucial for developing effective policies and mitigation strategies to address microplastic pollution in Canada's aquatic ecosystems (Hataley et al., 2023).

In conclusion, the effectiveness of these initiatives is often hindered by inconsistent implementation, limited geographical coverage, and a lack of integration across ecosystems. For instance, while marine environments receive substantial focus, terrestrial systems like soils, critical to agriculture and food security, remain underrepresented in monitoring programs. Additionally, the reliance on regional frameworks may limit the global comparability needed to address this transboundary issue effectively. Moving forward, it is essential to bridge these gaps by expanding monitoring efforts, fostering stronger international collaboration, and ensuring that the findings directly influence tangible policy actions to mitigate microplastic pollution and its far-reaching environmental and health impacts.

8.5 KEY CHALLENGES

A key challenge in developing effective regulations is the lack of standardized measurement techniques for MPs in these matrices. This, coupled with a limited understanding of MP occurrence, degradation, fate, and potential health risks, hinders the establishment of such regulations (Raza et al., 2022, de Souza Machado et al., 2018a, Martindale et al., 2020). Ongoing research is crucial in addressing these gaps, focusing on improving analytical methods, assessing impacts on ecosystems and human health, and developing management strategies to mitigate MP pollution (Huang et al., 2021, Raza et al., 2022).

Regulatory frameworks addressing MPs remain limited, with most countries focusing on waste management and water treatment policies rather than explicitly targeting soil or groundwater contamination. This leaves terrestrial environments insufficiently protected from MP pollution. While some specific regulations targeting MPs are beginning to emerge, much of the work remains in its preliminary stages. However, there is growing recognition of the risks posed by MPs in terrestrial environments, and countries are gradually moving toward more comprehensive and targeted policies. Effective regulation needs interdisciplinary research and coordination among sectors. Reports from the United Nations Environment Program (UNEP) suggest that integrated approaches are essential to bridge regulatory gaps (UNEP, 2021).

9 REMEDIATION

9.1 DO TECHNIQUES TO REMEDIATING OR DEGRADING MICROPLASTICS IN SOILS ALREADY EXIST? AN OVERVIEW OF EXISTING METHOD AND THEIR TECHNOLOGY READINESS LEVELS (TRL)

Remediating MP from soil and sediment is challenging, but in recent years several promising techniques have emerged. However, each method has its advantages and disadvantages, and the overall effectiveness of soil MP remediation remains limited. The feasibility of remediating MP from soil depends on various factors, including the type and concentration of MP, soil characteristics, and the chosen remediation method. While complete removal of MP from soil is currently not achievable, several techniques show potential for reducing MP concentrations and mitigating their environmental impact.

Four remediation techniques have gained popularity for addressing the pollution by MP contaminants in different soil layers and sediments (Thapliyal et al., 2024). (1) Pyrolysis or photocatalytic degradation are effective ex-situ remediation strategies for MP treatment, particularly in the uppermost layer of soil. (2) Magnetic extraction offers a less destructive alternative to pyrolysis while maintaining similar efficiency, though it also remains an ex-situ remediation approach. (3) Phytoremediation involving plants and associated soil microorganisms proves a practical in-situ solution for the superficial soil layer, primarily in the vicinity of plant roots. (4) Microbial degradation is a suitable and in-situ method for addressing MP contamination in the subsurface layer of the soil (Zhao and Zhang, 2023). Table 15 presents the mechanisms, advantages and drawbacks associated with these five remediation technologies and technology readiness levels (TRL).

Table 155 | A summary of the advantages and disadvantages of the remediation technologies and their principles. Technology Readiness Level (TRL) TRL 1: Basic principles observed and reported. TRL 2: Technology concept and application formulated. TRL 3: Proof of concept demonstrated through experimental evidence. TRL 4: Component and/or system validation in a laboratory environment. TRL 5: Validation in a relevant simulated environment. TRL 6: Demonstration of a system or prototype in a relevant environment. TRL 7: Prototype demonstration in an operational environment. TRL 8: System completed and qualified through testing and demonstration. TRL 9: Actual system successfully deployed in an operational environment.

Technology	Mechanism	Advantages	Drawbacks	TRL level	References
Pyrolysis	Degrading long chain polymer molecules into smaller, simpler molecules by heat and pressure.	Pyrolysis products do not require treatment nor pollute water Easy and flexible handling process Low labor costs	Destruction of soil structure High temperature conditions required Other plastic polymers may potentially be formed through the reaction Expensive	TRL4	(Ni et al., 2020, Sharuddin et al., 2016, Yansaneh and Zein, 2022)
Magnetic extraction	Attaching magnetic particles to the surface of MP, followed by application of a magnetic force	High efficiency Material reusability.	Magnetic particles face challenges in selectively and stably adsorbing to MP	TRL3	(Liu, Under review, Ramage et al., 2022, Shi et al., 2022c)
Phytoremediation and immobilization	Phytoextraction Phytostabilization Phytofiltration	Environmentally friendly	Limited effectiveness for larger-sized MP Plant species and their growing needs need to be considered; Longer remediation cycle	TRL2	(Sarwar et al., 2017, Singh et al., 2022, Ting et al., 2018)
Microbial degradation	Biodegradation Biofragmentation Assimilation Mineralization	Environmentally friendly Cost-efficient	Difficult to identify and isolate highly active and functional microbial consortia.	TRL6	(Omidoyin and Jho, 2023, Tiwari et al., 2020, Yuan et al., 2020)

For the **pyrolysis** of MP in soil or sediment, early studies primarily utilized Py-GC/MS for direct measurement of MP within the matrix (Hermabessiere et al., 2018). Subsequently, researchers investigated the effect of pyrolysis on MP reduction in sewage sludge and soil through a lab-scale study (Ni et al., 2020, Hu and Jiang, 2024). Micro-Raman analysis showed that microplastic concentrations in sludge residues significantly decreased from 550.8 to 960.9 particles/g to 1.4–2.3 particles/g as the pyrolysis temperature increased to 500 °C, with no small MPs (10–50 µm) remaining. And the experimental data showed that Polyethylene-based and polyvinyl chloride-based MP-contaminated oil-soil can be fully remediated at 500°C through rapid pyrolysis within 15 minutes (Hu and Jiang, 2024). However, pyrolysis causes the most destruction of soil structure among all methods. Due to the high energy costs required for this remediation method and the need to prevent the formation of other polymers in the process, this approach remains at the laboratory validation stage (TRL 4).

Magnetic extraction is a technique which removes MPs from soil by attaching magnetic particles to the surface of the MPs, followed by the application of a magnetic force to separate them. This method is known for its high efficiency and offers the advantage of material reusability, however, this is only confirmed in lab settings. Their practical application in larger soil environments faces significant challenges.

Phytoremediation offers potential through root immobilization or extraction, as small MP and certainly NP can be absorbed by plant roots and transported to aboveground tissues (Zhang et al., 2023b). Root immobilization reduces MP mobility and bioavailability, thereby mitigating ecological risks, although it does not eliminate MP from the environment (Li et al., 2024). In contrast, plant-based extraction effectively removes MP by incorporating them into plant tissues, with certain species (Austen et al., 2022), such as woody plants and duckweed (*Lemna* minor), demonstrating significant accumulation potential (Austen et al., 2022, Kalcikova et al., 2017).

Plant uptake of MP is limited by particle size, with only smaller MP capable of being absorbed by roots. Several studies have reported the uptake of different polymer MP (PE and PS) ranging from 0.1-10 µm in size by various crop plants. This was observed in wheat (*Triticum aestivum* L.), maize (*Zea Mays*), and carrot (*Daucus carota*) (Li et al., 2023b, Dong et al., 2021, Li et al., 2023c). In the future, it is essential to consider the selection of appropriate plant species that are economically viable, considering their specific growth requirements, and the extended timeframes needed for remediation. Currently, phytoremediation and immobilization have been predominantly validated in aquatic environments (Li et al., 2023c)), with existing studies merely demonstrating the potential of certain terrestrial plants for these capabilities (TRL 2). The selection of plants also comes with several requirements. Remediation plants need to adapt to local growth conditions and cannot be edible; Recovery and management of plant residues after remediation should prioritize plants that can be burn at very high temperature for purposes such as energy production. Overall, for soil remediation, further laboratory validation is required.

Microbial degradation is an eco-friendly, and cost-effective approach to address MP contamination (Velez et al., 2018). This process involves biochemical pathways, influenced by factors such as microbial activity, environmental conditions, and the physicochemical properties of MP (e.g., hydrophobicity, additives, and surface persistence) (Velez et al., 2018). Key mechanisms include biodeterioration, bio-fragmentation, assimilation, and mineralization, where microorganisms break down complex polymers into simpler forms, eventually converting them into CO_2, CH_4, and H_2O. Microbial degradation can be enhanced through bio-stimulation (providing nutrients) and bio-augmentation (adding capable microorganisms). However, challenges remain in understanding microbial attachment to plastics and the formation of biofilms by complex microbial communities including bacteria, archaea, fungi, viruses, and protozoa, which is referred to as the plastisphere (Joos and De Tender, 2022). The plastisphere plays a critical role in the degradation process.

Microbial degradation occurs in both terrestrial and aquatic environments and can degrade different types of MP. Table 16 presents microorganisms identified to degrade specific polymers. Besides direct degradation of polymers, several enzymes are also identified as the catalyst required for the degradation of complex polymers such as polyurethanase, esterase, laccases, hydrolases, dioxygenases, and peroxidases (Tournier et al., 2023, Amobonye et al., 2021, Santacruz-Juárez et al., 2021).

Table 16 | Microorganisms identified which degrade specific polymers. Table adapted from Thapliyal et al., 2024

Technology	Microorganisms identified to degrade polymers	Advantages
PE	*Nesiotobacter, Achromobacter, Micrococcus, Aspergillus, Acinetobacter, Staphylococcus, Arthrobacter, Bacillus, Stenotrophomonas, Comamonas,* and *Pseudomonas*	(Rambabu et al., 2023)
PET	*Ideonella, Aspergillus, Bacillus, Staphylococcus,* and *Saccharomonospora*	(Samak et al., 2020, Tournier et al., 2023)
PVC	*Pseudomonas* and *Achromobacter*	(Rambabu et al., 2023)
PU	Fungi: *Chaetomium, Curvularia,* and *Aspergillus* Bacteria: *Corynebacterium, Acinetobacter, Pseudomonas, Penicillium, Geomyces* and *Mortierella*	(Khandare et al., 2022)

Comprehensive analysis of MP degradation mechanisms, including changes in physicochemical properties and cellular responses, is crucial for advancing this remediation strategy.

Although microbial degradation is in theory an interesting technique for bioremediation, it still faces many drawbacks. First, it should be noted that although several organisms have been reported that can degrade plastics, that for most of these organisms it is unclear if they mineralize the plastic (up to CO_2 and H_2O) or merely fragment the plastic into smaller fragments. For most of the studies, microbial degradation is measured through the release of CO_2, while this is not trustworthy and can also be the result of fragmentation in smaller microplastic fragments. Second, in most cases the degradation rate is very slow, for which a complete conversion of the microplastic fragments can take up to multiple years.

Additionally, in recent years, a variety of emerging technologies (e.g., membrane filtration, chemical induced coagulation-flocculation-sedimentation, adsorption) have been developed for the efficient removal of MP in water (Lu et al., 2023). Membrane filtration is one of the most commonly used methods to remove MP, which allows for precise separation of different types of MPs in a facile, efficient and environmentally friendly manner. Chemical induced coagulation-flocculation-sedimentation (CFS) is also a representative method for removing MP, mainly includes three processes: coagulation, flocculation and sedimentation. Coagulation is the process of aggregation of colloidal particles and tiny suspended matter in water induced by adding a certain amount of coagulants, which is a widely employed, highly efficient, and cost-effective process in wastewater treatment. A recent study evaluated the effectiveness of modified starch and traditional coagulants (polyaluminum chloride and polyacrylamide) in coagulation, followed by biochar filtration (Tang et al., 2024). While coagulation is cost-effective and simple, residual coagulants may pose ecological risks (Hoang et al., 2025). Adsorption is another commonly used method for removing MPs, which can be classified into two categories, i.e., physical adsorption and chemical adsorption. Biochar effectively removed over 90% of MPs from coagulation effluent, and the combined coagulation–filtration process achieved a 97% removal efficiency (Tang et al., 2024). Biochar and other nature-based materials are promising, environmentally friendly adsorbents for microplastic removal (Hoang et al., 2025).

9.2 PLASTICS RECYCLING AS SOLUTION TO MICROPLASTICS INPUTS IN TERRESTRIAL ECOSYSTEMS?

Plastic recycling involves a series of modifications and transformations required to recover feedstock from previously processed polymers, allowing them to be reused (Beghetto et al., 2021). The two main types of plastic recycling are **mechanical recycling** and **chemical recycling,** which are classified in primary to quaternary processes (Figure 19). Primary recycling is the recycling of high quality (mostly pre-consumer) waste into products with similar characteristics as the original product. Secondary recycling refers to the mechanical recycling of post-consumer waste, resulting in products with reduced quality compared to the original. Chemical recycling or tertiary recycling refers to advanced methods to depolymerize and recover hydrocarbon products from the waste plastics, mostly through thermal processes (mostly pyrolysis and gasification, followed by condensation of the pyrolysis products into different recyclable fractions) but also through chemical depolymerization (Hong and Chen, 2017). Quaternary recycling involves burning plastic waste to recover energy. Plastics that are sent to a landfill lose their value and become waste (Figure 19).

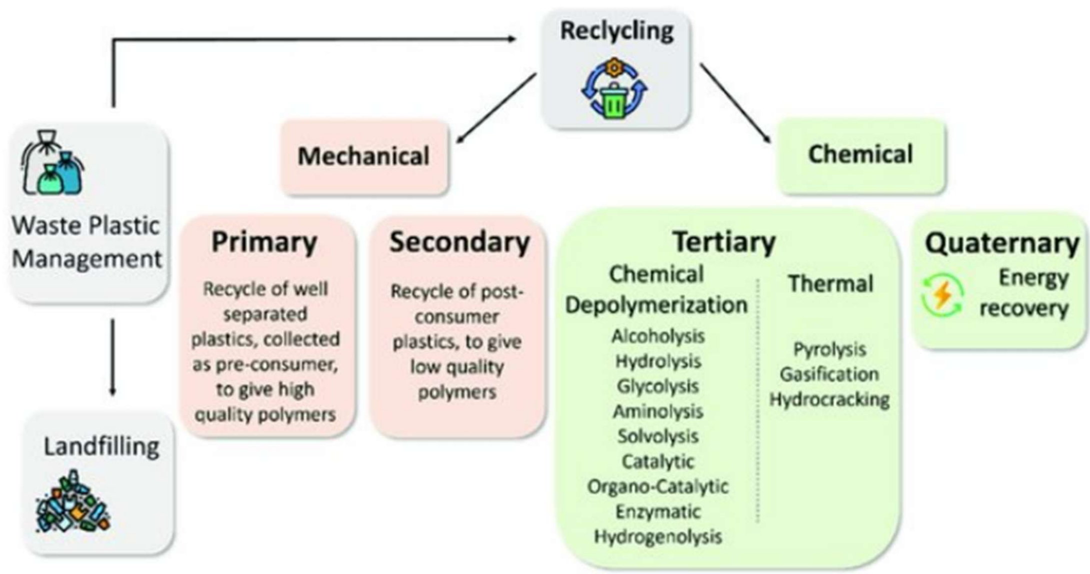

Figure 19 | Overview of plastic recycling techniques. Source (Beghetto et al., 2021).

Plastic waste recycling is currently dominated by mechanical recycling, with PE and PET being the most recycled post-consumer plastics globally according to this pathway (Singh et al., 2024) (Figure 20). Mechanically recycled plastics tend to continuously degrade in the process and thus cannot retain their quality after one or more recycling loops. This is especially problematic with respect to food-grade and contact-sensitive plastic applications (Rizos et al., 2023). There is certainly potential for mechanical recycling to be further improved pending innovation and following increased awareness about selective collection and sorting, but even then, it will not be possible to treat all waste streams via mechanical recycling, in particular for sensitive applications that require high output quality. More recently chemical recycling has been put forward as a means to recycle plastics a near infinite times. However, to date chemical recycling has been implemented at industrial scale to a very limited extent only because the process is economically inefficient (requirement of high temperature or pressure, additional costs for collection, washing, and sorting steps, and the relatively very low cost of raw materials used in plastic production) (Schade et al., 2024).

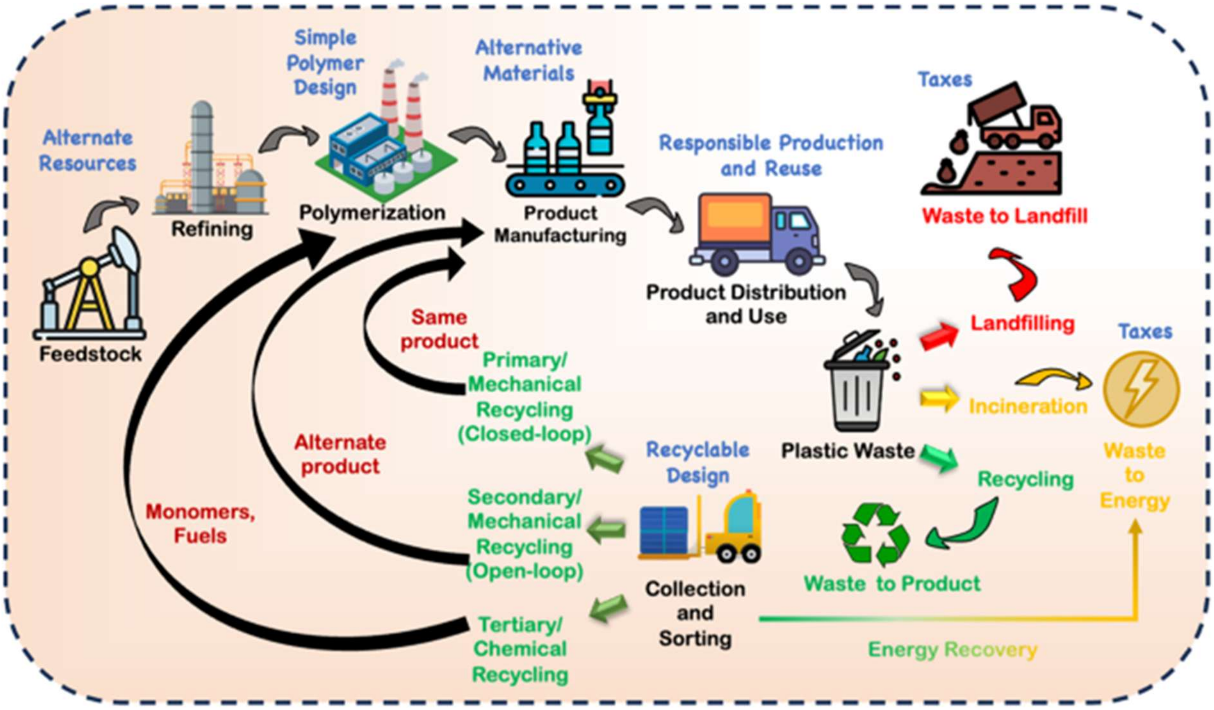

Figure 20 | Schematic of the plastic life cycle (black), different plastic waste handling methods (landfilling, incineration, and recycling), approaches to recycling (green), and solutions to achieve sustainability (blue). Source Singh et al. (2024).

Plastic recycling is generally viewed as the most efficient way to reduce environmental impact of plastics production and use, including reducing MPs in the environment. Recycling allows to better close the materials and energy cycles, resulting in important reductions in greenhouse gas emissions (Saleem et al., 2023), and reduces the amount of plastic waste going to incineration, landfills, or worst case directly to the environment. However, this view has been challenged by numerous studies that point out the disadvantages and environmental risks of recycling. Problems identified with current plastics recycling, which currently on industrial scale is nearly always mechanical, include:

- The limited range of plastic types that can be effectively recycled.
- Significant losses of MPs into the environment during the recycling process.
- Rapid deterioration of plastics quality, strongly limiting the number of recycling cycles.
- Accumulation of toxins in recycled plastics, posing health risks to both workers and consumers.
- An often-overlooked issue, which we identify as a significant problem, is that recycled plastics are frequently repurposed into applications that are notorious contributors to environmental microplastic pollution.

The following sections will delve deeper into these problems.

9.2.1 Limitations set by plastic types

While recycling is widely touted as the pathway to follow towards more sustainability, global recycling rates in 2019 remain extremely low at only 9% (Singh et al., 2024). In Europe, recycling rates rose up to 41% in 2022 (Eurostat, 2024). One of the main barriers towards greater plastics recycling rates is the limitations set by the type of plastics. Recycling is largely limited to thermoplastics, whereas thermosets, where the covalent crosslinks prevent remelting even at high temperatures (Ignatyev et al., 2014), can be ground into fine powders that can only be used for certain downgrade applications. Similarly, elastomers (tyres) have a very uncertain recycling fate (e.g. (Garcia and Robertson, 2017)). Increasingly plastics are also being used in all types of composite materials (mixed with fiberglass, carbon fibers, wood, …) conferring specific desired properties to the plastics, but resulting in substantial separation hurdles.

9.2.2 Losses of microplastics during recycled plastics production

Plastics recycling plants have been shown to release large quantities of MPs into waste waters and the environment during the actual recycling process. The size reduction phase, involving the mechanical shredding of the plastic waste, has been identified as predominant source of microplastic generation, with plastic type significantly affecting microplastic generation (MPs generation rate well correlated (R^2 = 0.88) to the hardness of the plastic) (Stapleton et al., 2023). Brown et al. (2023a) studied a mixed plastics recycling facility in the UK and found that filters installed to remove the MP from the plants' waste waters were highly efficient in removing the larger (>40µm) MP fractions, but were totally ineffective for removing small (<5µm) MP, with estimated yearly MP discharges of between 59 and 1 184 t.

9.2.3 Limited number of recycling cycles

Currently most plastics can be recycled only a few times before they become plastic waste (Walker and Fequet, 2023). However, recycling facilities most often convert plastics into fabric materials, which are then repurposed into products such as clothing and shoes. Once these items reach their end-of-life, they are unlikely to be recycled again and typically end up in landfills or incineration.

In chemical or tertiary recycling, the plastics are depolymerized by pyrolysis, gasification or chemical treatment to yield plastic oligomers and/or monomers, resulting in pure hydrocarbons as feedstock for new plastic production. In theory, this process allows for endless recycling of the plastic polymer fractions. However, chemical recycling has failed to take off at industrial scales because of significant limitations including high energy consumption, financial viability issues, and environmental concerns. Additives such as plasticizers, fillers and colorants end up in the heavy, bitumen-like fraction that can only be used in construction, posing serious contamination risks.

An obvious limitation to recycling is the use of plastics in applications where those can practically not be recovered anymore, such as in construction works (plasticizers of concrete, plastic reinforced concrete, geotextiles, wood plastic composites, ...) and many outdoor applications.

9.2.4 Health risks

There are a number of reasons why recycled plastics may pose environmental and health risks that are potentially (much) larger than those of virgin plastics.

When hazardous chemicals such as phthalates, bisphenols, UV stabilizers, flame retardants and pigments are used in the production of the virgin plastics, these chemicals are likely to end up also in recycled plastics (Undas et al., 2023). Plastics can absorb contaminants both through direct contact and through absorption of volatile compounds (Cook et al., 2023). Plastic containers may hold cleaning solvents, pesticides and other toxic compounds, residues of which during recycling will be introduced as additional contaminants to those already present from the virgin plastics production process.

New toxic chemicals can be generated unintentionally upon heating of the plastic waste during the recycling process, and these may end up in the recycled plastic product. Mechanical recycling of PET plastic in samples with very low rates of PVC contamination led to the formation of benzene that was found in the recycled plastic product (Brouwer et al., 2020). Recycling of plastics containing flame retardants seems to be particularly problematic (Pivnenko et al., 2017). Bromophenol concentrations of 26 000 000 ng/g (2.6% w/w) of Tetrabromobisphenol A were measured in a sample of recycled acrylonitrile-butadiene-styrene. The presence of selected brominate flame retardants in food contact materials and children's toys purchased on the European market was suggested by Puype et al. (2015) and Samsonek and Puype (2013) to be the result of contamination of the polymer products with recycled plastics of waste electrical and electronic equipment (e-waste). Sorting challenges and the presence of certain packaging components in sorted materials can also lead to toxicity in recycled plastic.

The health risks are highest for plastics recycling workers and communities nearby recycling plants, often in developing countries, as demonstrated in numerous studies. However, there are also well documented health risks for the end user of recycled plastics, in particular when these are used for food packaging, toys and other applications leading to direct oral or skin exposure. While health risks also apply to the production, the use and contact with virgin plastics, the risks with recycled plastics are much less under control because of the reasons given above.

9.2.5 Low value applications accelerating MP contamination

Recycled plastics are of inferior quality as compared to virgin plastics, and are often used for clothing, shoe wear and many (low grade) outdoor applications (pavement, playground equipment, poles, benches, …) for which non-plastic alternatives could easily be used. However, these applications are exactly the ones that are associated with production of large amounts of MPs, due to their direct exposure to intensive abrasion and/or to the impact of weather conditions (heat, cold, water, UV radiation) and in that sense, recycling plastics likely greatly accelerates microplastic pollution of terrestrial environments. This means that, apart from the health risks, the impact of recycling on MP pollution largely depends on the nature of the resulting product (packaging, bottles, bags, … versus clothing, outdoor applications, …).

At present plastics recycling is practically synonym for mechanical recycling. While mechanical plastics recycling does aid to better close the material and energy cycling, it does not contribute to solving the problem of MP pollution, rather on the contrary. There are numerous practical problems associated with plastic recycling that put limits on the further expansion of this type of plastics waste management. There are numerous environmental and health risks associated with this type of recycling. Because of the typical applications for which recycled plastics are being used (textiles, outdoor applications), they strongly contribute to MP inputs in soils, surface waters and sediments.

PART B | MICROPLASTIC PROBLEM AND RISKS

1 ESTIMATE EXTENT OF MICROPLASTIC PROBLEM AND RISKS

In this second section of the report, we synthesize insights from PART A to examine the impact of MPs on soil, groundwater, and sediment, with a focus on currently available data and risk assessments. This section also addresses exposure to microplastics in relation to existing knowledge on ecological and human toxicological risks. As highlighted in the knowledge overview (PART A), microplastics are pervasive across all environments — from soils and groundwater to marine ecosystems — and have even been detected in remote natural areas with minimal human activity.

Despite their widespread presence, the question remains whether MPs pose a significant risk to: **(a) soil ecosystem functioning (related to soil quality measures including the soil microbial community, microfauna and nutrient cycling); (b) plant health and development; (c) the environment; and (d) human health**. Identifying the risks of micro- and in extension also nanoplastics forms the basis for regulations and the design, prioritization and timing of solutions (see PART C).

In recent years, scientists have made significant progress in developing risk assessment frameworks for MP particles (Koelmans et al., 2020, Koelmans et al., 2023, Gouin et al., 2019). The main criterion for the validity of the risk assessment framework is that it contains an exposure, a hazard and a risk characterization component, and that it has **sufficient accuracy and completeness to estimate the complexity of environmentally relevant microplastic mixtures**. This clearly showcases the complexity of the problem, as multiple components or variables should be included for an ecological, but also human health, risk assessment. Unlike "classical" chemical contaminants, which have a clearly defined chemical structure, MPs are complex mixtures of polymers with fillers and additives in all possible combinations and ratios, and change composition over time as the result of weathering and the sorption/desorption of compounds. In addition to the physical properties of microplastics (microplastic concentration, polymer composition, shape, and size distribution), Koelmans et al. (2023) defined 15 other components which should be considered, including several parameters discussed in Part A, such as additives, sorbed chemicals and particle bioavailability (Table 17). Without going into detail, currently two risk assessment methods exist (PDF: Probability Density Functions; BR: the Bucci and Rochman framework (Bucci et al., 2022)), which take at least more than 3 of these components into consideration, thereby moving away from the simpler risk assessment methodologies often focusing merely on microplastic concentration.

For each of these parameters it is not clear which will be the most determining regarding environmental and/or health risks, and therefore all should be considered. Nevertheless, specifically for soil-related ecosystems, this has proven to be difficult as the number of studies is limited compared to other environments (e.g. marine, freshwater).

Despite these limitations, this section aims to outline both the current and potential future risks of microplastic pollution in soil-related ecosystems. This will be done through discussing the four physical properties (concentration, size, polymers and shape) of MPs that affect their environmental response and toxicological effects.

Table 17 | Components of a risk framework assessment. Source Koelmans et al. (2023).

Essential Component	PDF Framework	BR Framework
1. Characterisation of physical properties [a]	Lossless probability density functions (PDFs), applicable to all possible characteristics	Simplifying categories, i.e., 6 for size, 3 for shape, 5 for polymer
2. Extent to which the entire microplastic continuum is covered [b]	PDFs cover the entire continuum, e.g., for sizes from 1 to 5000 µm, regardless of analytical limitations	Covers the particles in a sample, which is thus limited by the analytical method that happens to be used
3. Representativeness of the scale of the assessment [c]	Allows probabilistic system-wide extra- and interpolation while taking system dynamics into account	Limited to the scale of 'snapshot' samples that are assumed to be location-specific
4. Additives [d]	Continuous dose-response relationships, accounts for all chemical exposure pathways, uses PEC/PNEC approach	Simplified to three exposure categories, considering only possible exposure via microplastic, ignoring other routes of chemical exposure
5. Sorbed chemicals [d]	Continuous dose-response relationships, accounts for all chemical exposure pathways, uses PEC/PNEC approach	Simplified to three exposure categories, considering only possible exposure via microplastic, ignoring other routes of chemical exposure
6. Chemical exposure scenario [e]	Actual environmental concentrations to approximate the situation in nature	Concentration in the original product
7. Particle bioavailability [f]	Particle size versus organism mouth opening or translocation barrier	Not accounted for
8. Effect assessment [g]	Effect thresholds from standardized tests, combined in e.g., SSDs	Not accounted for
9. Strategy regarding particles to be tested [h]	One environmentally relevant polydisperse mixture of particles, reducing the need for alignments	Sequential testing of many monodisperse particle types, dissimilar to environmental mixtures
10. Species specificity [i]	Through species specific bioavailability and -sensitivity to particle and chemical effects	Not accounted for
11. Adaptation to habitat type j	Habitat specific SSD	Not accounted for
12. Risk characterization [k]	PEC/PNEC for toxicologically relevant metrics that are motivated from known effect mechanisms	Not accounted for
13. Consistency with known effect mechanisms [l]	Recognizes the food dilution mechanism, and mechanisms triggered by translocation. Quantitative	Agnostic, qualitative
14. Coherence with risk assessment in existing policy frameworks [m]	Complies to the ruling risk assessment paradigm	Not coherent. [n]
15. Availability of open science tools [o]	Accessible to a wide audience through the ToxMek web application	Not available
16. Degree of acceptance and integration in science and policy [p]	Implemented in a risk management framework and regulation for California. This included an expert elicitation regarding the validity of the concepts and outcomes of the assessment. Used in five scientific studies	Not yet implemented or used elsewhere

a	Needed to be able to quantify bioavailability and toxicity caused by characteristics.
b	Needed to assure no relevant fractions are overlooked. Only if the naturally occurring extremes for all relevant environmental characteristics of microplastics are covered, can the framework be said to maintain the complexity of microplastics.
c	Needed to ensure that the spatial scale matches that of communities to be protected, and that spatiotemporal scales take into account the variability of exposure concentrations caused by hydrological dynamics in aquatic systems.
d	Needed to address the contribution to effects caused by additives and sorbed chemicals. Following established concepts in risk assessment science, exposure is expressed as a measure of Predicted Environmental Concentration (e.g., PEC), whereas Predicted threshold No-Effect Concentrations are referred to as PNEC. Chemical risk characterization is quantified by the PEC/PNEC ratio.
e	The exposure scenario should be environmentally relevant, i.e., reflect the characteristics of exposure as they would occur in nature.
f	Particles that are not bioaccessible and/or bioavailable should not be taken into account.
h	The risk depends on the sensitivity of the organisms to effects, which must therefore be taken into account. Species Sensitivity Distributions (SSDs) are often used to adequately protect the most sensitive species in the food web.
i	Impact assessment requires threshold effect concentrations for species, where the concentration refers to the complex mixtures of particles as they occur in the environment. Ideally, the effect concentrations thus relate to environmentally relevant mixtures of particles.
j	Since the effects of stressors such as micropolymer particles depend on species traits, the relevant traits must be taken into account. It should be recognized that species sensitivities can be habitat specific (e.g., freshwater, estuarine water, seawater, sediment, soil), therefore the effect and risk assessment should be as habitat specific as possible.
k	For risk assessment, a quantitative characterization of the risk (e.g., PEC/PNEC) should be provided.
l	The assessment needs to be consistent with known effect mechanisms so that the correct exposure and effect metrics can be selected. Only when done correctly can a meaningful and consistent risk characterization be obtained.
m	Deviating from existing and accepted (risk assessment) knowledge and concepts known to managers is not recommended if it is not actually necessary.
n	Reports a risk assessment framework of which the outcome ('risk of a sample'), is a hazard value. Biological relevance remains unclear.
o	As long as concepts and algorithms are only described in scientific literature, they can be difficult to access for users such as risk managers. User-friendly tools are therefore recommended.
p	Acceptance of a scientific theory, model or framework by scientists and managers is an important measure of the validity and value of those products. Older frameworks have an advantage on this criterion.

1.1 MICROPLASTIC RISKS RELATED TO CONCENTRATION

1.1.1 Ecological risks

For **soil ecosystems**, recently a first risk assessment has been conducted, making use of the PDF framework (Redondo-Hasselerharm et al., 2024). This data was based on 51 studies (up to publication year 2023), representing 241 measured environmental concentrations (MECs). The researchers took into account six threshold criteria: 1) the type of soil in the tests; 2) reporting of the effect threshold concentrations per mass of soil; 3) chronic exposure duration; 4) evaluation of endpoints that target effects at the individual level (e.g. survival, growth, reproduction); 5) detection of significant concentration-dependent adverse effects and 6) reporting of the size of the MP used in the tests. It thus needs to be noted that only a few of the components referred to in Table 1 were taken into account for this risk assessment, majorly because the other data was missing in the studies. The researchers specifically focused on the number of particles per kg dry soil and toxicity data for soil invertebrates (e.g. nematodes, earthworms) and plants. Therefore, this risk assessment is solely related to ecological risk rather than human health.

Overall, the researchers reported HC5 values – representing the concentration of MP particles that would protect 95% of evaluated soil and plant species - ranging between 4.0×10^7 and 2.3×10^8 MP particles (1-5 000 µm)/kg of dry soil for different MP sources. They recommend to utilize the lowest calculated HC5 value (4.0×10^7 particles/kg dry soil), which should, in principle, provide sufficient protection for all plastic sources examined. If we compare this to the currently found microplastic concentrations globally (Table 6), even the highest reported concentration of 1.2×10^7 particles does not reach this number. For the Benelux, the MiCoS project is currently studying the number of microplastics in arable fields. Of the 240 soils, from 82 we have at least one measurement, showcasing an average concentration of 3 530 MPs/ kg dry soil and a minimum concentration of 100 particles/kg dry soil and a maximum concentration of 60 100 MPs/kg dry soil, i.e. 3 orders of magnitude lower than the reported HC5. This research is, however, still ongoing and the values mentioned here will likely change by incorporating more measurements and by including a second measurement technique, with potentially exceeding the HC5.

A similar risk assessment has been conducted by the same group on **freshwater sediments,** with risks characterized for effects triggered by food dilution or translocation. There HC5 of 4.9×10^9 particles/kg dry sediments [Confidence intervals (CI): 6.6×10^7, 1.9×10^{11}] for food dilution and 1.1×10^{10} particles/kg soil [CI: 3.2×10^8 - $4{,}0\times10^{11}$] for translocation, respectively. Here data was obtained of 60 studies, reporting 103 MECs in total. The HC5 lower limit for volume and area was here however exceeded in 32% and 17% of the locations, showcasing that for sediments currently the risks might be higher compared to soil ecosystems. **These risks are expected to only increase as the plastic emissions and exposure to MP will only increase in the future.**

For **groundwater**, no risk assessments are currently present and therefore HC5 values cannot be given. Concentrations of MP in groundwater are on 1 up to 100 particles/L, but can exceed 2 000 particles/L. The currently observed microplastic concentrations in groundwater are unlikely to have direct ecotoxic effects, but might be an important MP source for freshwater ecosystems, agricultural land through irrigation or drinking water production thereby (re)entering the food chain.

Although these risk assessments are highly necessary and may form the foundation for future policies, several limitations exist, particularly in relation to the assessment of soil ecosystems. While this study is the first and only fully aligned ecological risk assessment specifically parameterised for microplastics in soils, it cannot be regarded as a truly global assessment due to a lack of data from many countries, including Belgium. This highlights the need for more spatially representative assessments to be conducted in the near future, which, once incorporated, may lead to different risk assessment outcomes. Second, most of the conclusions are related to agricultural land, as most studies (66.5%) published up to 2024 (Figure 10), were focusing on agricultural soils (arable and horticultural soil, greenhouse soil, plantations). Third, these ecological risk assessments are inherently uncertain (Redondo-Hasselerharm et al., 2024). It is therefore recommended to further quantify the uncertainties, specifically related to soil environments, in the future. More specifically: soil sampling depth and volume, the homo- (or hetero-)geneity of concentrations in samples, the ingestion of particles by invertebrates, translocation of microplastics in the plants, effects of microbial activity and many more should be taken into account. Fourth, methods now fall short in measuring the entire MP continuum and are often limited to plastic sizes > 20 µm, particularly for soil environments. Including empirical data for smaller particle sizes will improve the accuracy of the MP risk assessment. At last, this risk assessments takes into account only a few variables or parameters, mostly because of the lack of data regarding microbial communities on the plastics (and thus pathogens), chemical data, but also the use of different polymer shapes than spheres. To improve these risk assessment analyses, it is up to the scientific community, and to policy, to include this information in upcoming research and monitoring studies.

1.1.2 Human health risks

Human health risk assessment of MNPs remains in its infancy, despite evidence of their widespread occurrence and systemic bioavailability in humans. Research indicates that inhalation and ingestion are the primary exposure routes, with MPs detected across multiple organ systems, including the cardiovascular, digestive, respiratory, and endocrine systems (Chapter 5.2).

Current risk assessment frameworks for human health for MNPs face significant limitations across exposure, hazard, and risk characterization. Exposure assessment (either direct or indirect exposure) is hindered by the lack of standardized methods to quantify MNPs in environmental and biological matrices, particularly for nanoplastics (<1 µm), which evade detection due to analytical limitations in size resolution and polymer identification. Hazard characterization struggles with the multidimensional complexity of MNPs, which vary in polymer type, size, morphology, and additive/contaminant loads, complicating dose-response relationships. While frameworks like POLYRISK's Integrated Approach to Testing and Assessment (IATA) propose grouping MNPs by properties (e.g., OECD's *Polymers of Low Concern* criteria), uncertainties persist about whether *in vitro* or animal model outcomes (e.g., oxidative stress, inflammation) translate to human health effects. Risk characterization is further hampered by the absence of epidemiological data and health-based guidance values, leaving critical gaps in understanding thresholds for toxicity. Additionally, synergistic effects from adsorbed pollutants (e.g., heavy metals, pathogens) and bio-persistence of fibers are understudied, while methods to assess combined physical, chemical, and biological hazards remain fragmented. Current frameworks, though innovative, rely heavily on extrapolations from non-human studies and theoretical models, limiting their applicability to real-world exposure scenarios.

1.2 MICROPLASTIC RISKS RELATED TO SIZE

The effects of microplastics will be highly dependent on the size of the microplastic particle. As shown in Part A, the size will determine MP effects on uptake (by plants, invertebrates, but also humans), migration and potential effects on plant growth and the microbial community (Table 18).

Table 18 | Upper size limit where effects are detected for soil and plant species and humans.

Effect by microplastics on soil ecosystems		Upper size limit
Microplastic uptake soil biota	Earthworms	150 µm
	Nematodes	10 µm (20-30 µm for larger species)
Soil quality	Fastest migration through soil layers	10 µm
	Disruption soil aggregation	Particularly smaller (<50 µm) particles
	Alteration of soil structure	>100 µm
Microplastic uptake humans	Respiratory exposure	10 µm
	Dermal exposure	10 µm
	Ingestion	5 µm

Based on the overview in Table 2, the smallest microplastics are expected to have the most detrimental effects on both **soil ecosystem functioning** and **human health**. Analyzing the microplastic sizes reported in the 85 studies described in Part A (Section 3.2.1) reveals that only a minority of studies report a minimum microplastic size of **1 to 10 µm** (Figure 1). This underrepresentation is largely due to limitations in extraction technology and analytical methodologies, making the detection of the smallest microplastics particularly challenging.

Despite these limitations, research indicates that **smaller microplastics exhibit the highest ecotoxicological and human toxicological risks**. Consequently, their absence or underrepresentation in risk assessment studies suggests that these assessments may significantly underestimate actual risks (Figure 21). As analytical methods improve and smaller size fractions can be quantified more accurately, the HC5 threshold (hazard concentration for 5% of species) will likely be revised downward, reflecting the greater risk posed by these smaller microplastics.

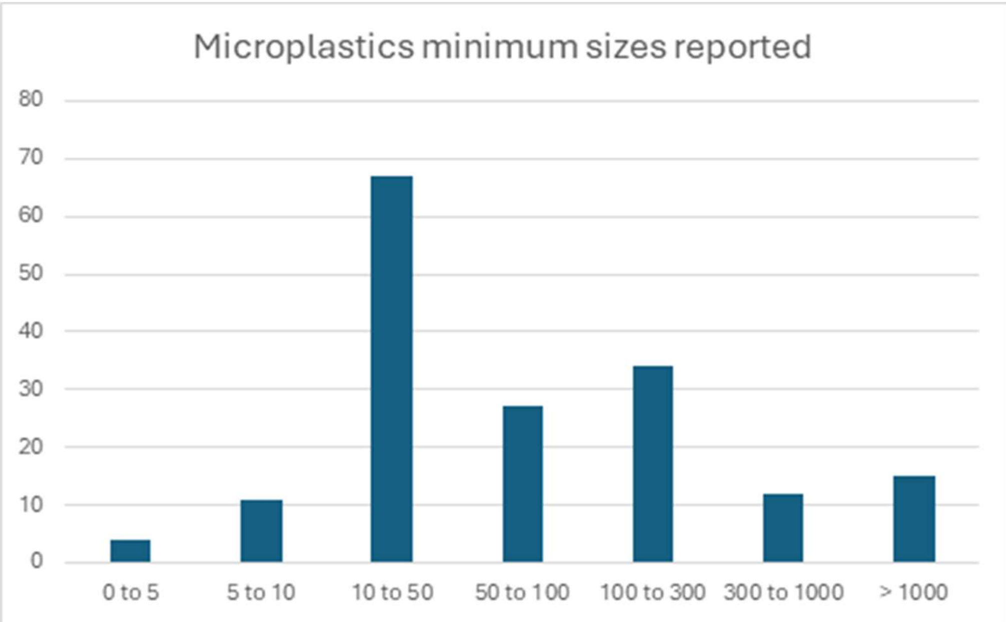

Figure 21 | Microplastic size distributions of the 85 papers described in Chapter 3.2.

1.3 MICROPLASTIC RISKS RELATED TO POLYMERS

The most produced plastics according to Plastics Europe in 2023 were PE (both high and low density), PP, PVC and PET, which are also the plastic types most often found as environmental plastics. Only a few studies have reported other polymer types.

During the interviews, several specialists indicated that the polymer type itself will be less decisive on ecotoxicity and human health effects, and that potential negative effects are particularly related to the particle itself.

Notwithstanding this observation by the experts, two remarks have to be made. First, one has to distinguish between the ecotoxicological and human health effects. While the particle itself will be the most determining factor for human health, degradation of plastics in soil will lead to the release of microplastics, but also the formation of other degradation products that can be harmful. Particularly for PVC, but also biodegradable plastics like PLA and PBAT, degradation products were shown to negatively impact plant growth and the soil microbial community (Liu et al., 2024). Therefore, polymer type will affect to some extent the efficacy of plastic on ecotoxicology, and should be included as one of the main variables in ecotoxicology studies. Second, most studies so far have been conducted on either PS or PE. PS is readily amenable for use in addition experiments. However, due to its relatively low production rate (Figure 3) and low occurrence in the studies on terrestrial ecosystems, it is perhaps one of the less relevant polymers. On the other hand, polyethylene is the most widely produced polymer and is widely used in agriculture. The relevance of this polymer is thus high, but most studies did not find detrimental effects of polyethylene use.

1.4 MICROPLASTIC RISKS RELATED TO SHAPE

Most of the research regarding effects of microplastics on the environment and human health has been done making use of spheres (also referred to as pellets, see Figure 2). Therefore, almost all data regarding risks represented in Part A and Part B is based on spherically shaped plastics. However, these particles are almost absent as plastics in the real environment, also because they degrade to fragments with irregular shape, or are present as fibers especially related to clothing.

It is expected that fibers have the most detrimental effects on the environment. For instance, according to De Souza Machado et al. (2018) MP fibers can effectively establish a tighter connection between microaggregates because of their linear shape, while MP fragments interact more loosely with the soil. In addition, fibers have been observed to decrease bulk density to a greater extent than beads and fragments (de Souza Machado et al., 2018a). Therefore, they will have a higher impact for affecting soil structure. In addition, fibers have a more even depth distribution, while e.g. films are more abundant in superficial soil layers (Hu et al., 2022). The difference in effects between fibers and other microplastic shapes in plant and soil health studies should be the topic of upcoming studies, as almost no information is available so far.

For **human health studies**, the effects of MPs on human health are still underexplored, and research focusing specifically on the shape of microplastic particles remains limited. Most available data is based on studies using spherically shaped MPs, often selected for their consistency and ease of use in laboratory settings. However, these shapes do not accurately represent the particles most commonly encountered in the environment. Irregular fragments, fibers, and films are far more prevalent in real-world exposure scenarios.

Importantly, it is currently assumed that health risks from microplastics in humans are a particle issue, rather than a polymer-specific issue. That is, the physical presence of particles—especially in certain shapes and sizes—is thought to play a larger role in triggering biological responses. Among these, fibers are of particular concern due to their widespread presence and their physical properties, which make them more likely to be inhaled or ingested. Synthetic fibers, such as those released from clothing during washing and drying, can become airborne and enter the human respiratory system. Their elongated shape and persistence may allow them to bypass the body's natural defence mechanisms, potentially leading to respiratory irritation, inflammation, or even chronic lung conditions.

The role of particle shape in crossing biological barriers, such as the gut lining or respiratory epithelium, is also a growing concern. Irregular fragments and fibers may interact more aggressively with tissues compared to smooth, spherical particles, potentially enhancing their ability to penetrate cells or accumulate in organs.

Given these considerations, there is an urgent need for future human health studies to broaden their scope beyond spherical particles and include a diversity of microplastic shapes—particularly fibers, which may pose heightened risks due to their shape, persistence, and real-world prevalence. Studies should also examine how these particle characteristics interact with chemical additives, environmental pollutants, and biological systems to better understand the full spectrum of potential health effects.

1.5 OTHER VARIABLES WITH HIGH RISKS

Whereas in the previous part we focused primarily on the risk introduced by the particle itself, it cannot be neglected that microplastics pose other risks as well, of which the leaching of chemicals (1) and the transport of hazardous chemicals or microbial pathogens (2) are probably the most important.

Yet, there is a consensus between experts that in terrestrial ecosystems these effects are less important than the effects of the MP given the mostly (extremely) low concentrations of both chemicals and pathogens and their strong attachment to the microplastic particle.

2 CONCLUSION

A number of parameters need to be taken into account to calculate the risks of MP (1) on the ecosystem or (2) on human health. Overall, it can be concluded that to advance future risk analysis studies, researchers need to report not only the microplastic concentrations, but also size distributions, the type of polymer, polymer shape and when possible, information regarding the chemical composition, added chemicals and even the biofilm formation (see Table 18). In this respect, the leading expert on MP risk assessments (Koelmans et al., 2023; interview) emphasized that we need to develop a framework that preserves the information on the complex nature of microplastics by capturing the hazard associated with both their physical and chemical characteristics to assess the risks.

In addition to this, we want to stress the importance of moving forward in the experimental set-up: there is an urgent need for more studies including microplastic mixes, including differences in polymer type, size and shape, in which environmentally relevant concentrations are used. This limits our understanding of microplastics risks tremendously, as in real-world environments complex polymer mixtures are present that may lead to different effects. In addition, future studies on ecotoxicology should include different soil types and plant species to capture the complexity of the terrestrial ecosystem.

PART C | FUTURE POLICY STUDIES AND POLICY SUPPORT RESEARCH

Microplastics research in soil and groundwater started less than a decade ago, and the scientific and methodological challenges, and the knowledge gaps, are numerous. Based on the extensive literature review and the risk analysis presented in parts A and B, respectively, we here summarize the main knowledge gaps which need to be addressed in order to come to an efficient and effective monitoring system, to mitigate risks, and to come up with effective measures to minimize or stop further MP inputs in terrestrial ecosystems.

1 KNOWLEDGE GAPS

1.1 SOURCES

The main sources of MP in soils, sediments and groundwater are still poorly known (PART A Chapter 2.2). The most important source of MP inputs is probably the simple consumer products that we use every day and that release MP very gradually, so that the MP production goes almost unnoticed (see also Chapter 9.2). This refers mainly to plastics used in consumer goods like carpets, curtains, boxes, shoe ware, clothing, etc. and outdoor plastics applications such as in plastic garden furniture, poles, gardening equipment, playgrounds, plastic paving, … Collectively, these are undoubtedly massive sources of MP production and pollution, but not considered in classical overviews of most important MP sources. As shown previously, these focus particularly on tyre particles, industrial pellets and macroplastics (with consumer waste products as the main sources within this category). There is thus an urgent need to clearly identify and quantify these overlooked MP sources. Once sources have been identified, the input pathways to soil and groundwater can be studied in a targeted way and addressed/mitigated.

When it comes to MP pollution, the focus is mostly on fossil fuel based MP. However, the question is to what extent biobased plastics are less harmful and accumulate less in the environment. Biobased is not synonym for biodegradable, and some popular biobased plastics are notably very resistant to (microbial) degradation and thus likely accumulate in soils. In addition, research has shown that some biobased plastics (PLA, PBAT) might be more harmful to the soil ecosystem and particularly plant development, leading to smaller plants, less fruits and retardation in germination. For natural ecosystems, but especially for agriculture, this could lead to major problems when these biobased microplastics accumulate in the soil. There should thus be more research into the environmental safety of biobased plastics as an alternative for fossil based ones.

In addition, we have the issues regarding the detection, quantification and risk assessment of rubber particles. Up to 20 distinct rubber types are on the market, with an additional variability due to proprietary formulations. While popular methods for microplastic characterization (µFT-IR, Raman spectroscopy) can identify plastic polymers very well, they face limitations for rubber. Carbon black’s high absorbance hampers FT-IR, while Raman spectroscopy long run time leads to slow analysis. So far, thermoanalytical techniques are preferred for the detection of rubber, but cannot report numbers of particles. This has led to a high uncertainty of the number and amount of rubber particles in the soil. Also the effects rubber has on the land is understudied and only a few articles have focused on the problem. Scientists should therefore address the issue, in combination with micropolymer particles, in future studies.

1.2 CONCENTRATIONS/BEHAVIOUR

One can only report what can be measured. Given the enormous methodological challenges, without doubt MP concentrations in soils, sediments and groundwater must be extremely underreported. The smallest fractions (<20 µm, but in most research <100 µm) by definition cannot be recovered and hence not reported, meaning that at this moment, we have no reliable data on MP concentrations in soils and groundwaters, especially when it comes to the small MP fractions. This (large) share of MP 'dark matter' presents a fundamental problem when threshold values need to be defined, given that a threshold would only make sense in case it can effectively be measured.

The extent to which MP accumulate pollutants (heavy metals, pesticides, other organic contaminants) and pathogens in soil and the nature of this accumulation (transient or persistent) remains highly uncertain, and would need systematic research into MP eco-corona dynamics.

1.3 ECOTOXICOLOGICAL AND HUMAN HEALTH RISKS

From the literature, one can conclude that at current concentrations, the expected impacts of MP on soil biota and on soil functioning are limited, and probably not measurable. Research should further focus on how soils, sediments and groundwaters regulate the transfer of MP to other ecosystem compartments, namely atmosphere, surface water and biosphere, and put risks to human health.

There is not much knowledge on which microplastic properties are most determining ecotoxicological and human health effect: is it the polymer type, polymer size, polymer shape, the additives, the nature of the eco-corona, or combinations of these? The uncertainties surrounding this greatly complicate all efforts of risk inventorisation, because it is unclear which MP properties are most important to analyze and report. There is a risk that current inventories fail to report specific MP properties that later turn out to be crucial. More information from such ecotoxicological/human health risks is thus urgently needed, to make sure that we report the relevant MP properties.

There is only very fragmented knowledge on the ecotoxicological and human health risks of plastic additives. Currently substances can be used in manufacturing processes, until a serious environmental risk is identified, but at that time the pollution has taken place already and may prove to be practically irreversible. There is a list of 16 325 chemicals potentially used or present in plastic, of which only 6% are currently internationally regulated. More than 4 200 of these plastic chemicals are of concern because they meet one or more criteria of PBT or PMT. So rather than waiting for the problem to manifest itself, regulations should be proactive and not allow the use of suspect chemicals until it is proven that they are effectively harmless to the environment.

Ecotoxicological and human health risk assessment almost necessarily look at a limited number of factors (such as one plastic type, one exposure route, etc.) in order to keep experiments manageable. However, humans are exposed continuously to complex mixtures of MP and their additives, for which conventional toxicological studies cannot provide answers. Given the relatively recent nature of the research into MP health risks, long term monitoring of human health effects should be started or continued.

1.4 THRESHOLD VALUES

The uncertainties in which MP properties determine human health and ecotoxicological effects, and the difficulties in quantitatively recovering MP from soils, sediments and groundwater make any effort of establishing threshold values for environmentally safe concentrations highly speculative. Perhaps as a proxy, research can focus on correlations between (practically) measurable (threshold) MP concentration and the overall MP concentration in these matrices. Another fundamental problem is that threshold values likely depend on polymer size and shape, and that such values may need to provide information on all these aspects (number and size distribution of particles, fraction of fibers-pellets-flakes-…,) This information is unlikely to be delivered by conventional measurement techniques, so research should focus on new ways of MP characterization providing overarching insights in all these aspects.

1.5 REMEDIATION

Given the enormous variation in composition of MP and the continuous changes they undergo, remediation of MP pollution in soils, sediments and groundwater clearly presents a massive challenge. It will be difficult to find a correct balance between ecotoxicological risk and remediation efforts/costs. With the current state of knowledge, most should be expected from phytoremediation and microbial degradation. Therefore, more research is needed into the true potential (and possible risks) of these remediation strategies.

2 ADVICE FOR POLICY

Despite the limitations of the current technologies, we stress that in order to (1) determine human health risks; (2) effects on plant development and (3) leaching to the environment, we need information on the current pollution levels. This information is necessary in first instance to inform policy makers about regulations and legislation of (micro)plastic.

To make this step possible, policy makers, in collaboration with scientists, should focus on three actual problems regarding microplastic measurement: the methodology, the monitoring itself, and how to report.

Regarding the **methodology** we still face practical issues. Although initiatives are taking place (e.g. CEN TC 444 WG 6 at European level; INSOP (international network on soil pollution) on scientific level) to create standard methodology to measure microplastics in soils, all experts agree that we still have a long way to go. The current technology faces limits, including measuring only bigger sizes of microplastics (>20 µm) and the low detection limit of certain products (including rubber). In addition, the extraction of microplastics out of a soil compartment is done differently within each lab. In our opinion, this is where the first important steps could be made. As scientists we have the knowledge to extract microplastics from the soil and with the collaboration with policy makers, standardization among laboratories is possible. This discussion should be started as soon as possible and can be done by bringing together the main groups in microplastic extraction technology, for now situated in three European projects: MINAGRIS (https://minagris.eu/), Papillons (https://www.papillons-h2020.eu/) and MiCoS (http://www.micos.ugent.be).
While a standard methodology is necessary, we should however not wait to start setting up (smaller, particularly regional) monitoring campaigns. These monitoring campaigns are necessary to address eventually the risk on human and environmental toxicology. Once the methodology is in place, this should be regulated on European level. We would not limit monitoring to soil, but include both the surrounding environment (e.g. rivers, groundwater, air) to detect sources but also transmission, and food products (e.g. crops) as these are the direct link to human health.

While European legislation might not be possible in the near future, Belgium can be one of the pioneers regarding monitoring micro- and macroplastic pollution in terrestrial ecosystems. We can make use of current existing networks such as Cmon and LUCAS to identify and select sampling sites. The benefit of making use of these existing networks is that detailed information is already gathered related to soil physicochemical composition and microbial soil profile, which can be later on related to the plastic concentration. Other sampling sites can be included as well, which can be determined in collaboration with organizations such as ILVO (agricultural sites), INBO (natural regions) or the provinces (related to industrial locations/train stations/shoulders of highways/etc.).

Ideally, soils are monitored with a return period of 3 to 5 years, to detect microplastic accumulation over time. Based on the MiCoS protocol, and taking into account the high variation in soil nutrient and microbial profiles, we would advise having at least three sampling points per sampling site. Ideally, measurements are taken at different soil depths (e.g. 0-10; 10-30; 30-60 and 60-100 cm) but when limited in time and resources, we would focus on the top soil layer (0-30 cm) as this contains the highest microbial diversity (and activity) and is utmost important for plant growth. During sampling, strict procedures need to be followed to prevent contamination.

First, sampling material and clothing needs to be plastic free. If this could not be reached, including a procedure blank will report the contamination induced by the sampling procedure.

To end, policy can have a main influence on how we **report the data**. Depending on the measurement technique microplastic contamination will either be reported as number of particles (Raman spectroscopy, µ-FTIR, microscopy) or in mass (GC-MS). Both have pros and cons and in the most ideal situation, the report would contain both measures. If only one can be measured, the research question will determine the method of choice.

Overall, we can conclude that research regarding microplastic pollution in terrestrial ecosystems is getting the attention it needs, and that new research articles are appearing continuously. The implementation of monitoring campaigns, advancements in microplastic measurements techniques and experiments taking into account relevant microplastic concentrations will advance the field even further and will be the basis for regulations and legislations regarding future plastic use.

SUPPLEMENTARY

Table S1 | Supplementary table presenting the average microplastic count (particles/kg soil) along with corresponding references used for generating all graphs in Chapter 3.2.

Index	Continent	Avg. MP number (particles/kg soil)	Avg. MP Conc. (g/kg soil)	Land use	Reference
C1	Europe	NA	94.98333333	Agricultural	(Mengistu et al., 2022)
C10	Europe	104000		Agricultural	(Leitao et al., 2023)
C10	Europe	55000		Agricultural	(Leitao et al., 2023)
C10	Europe	92000		Agricultural	(Leitao et al., 2023)
C10	Europe	158000		Agricultural	(Leitao et al., 2023)
C10	Europe	92000		Agricultural	(Leitao et al., 2023)
C10	Europe	126000		Agricultural	(Leitao et al., 2023)
C10	Europe	127000		Agricultural	(Leitao et al., 2023)
C10	Europe	150000		Agricultural	(Leitao et al., 2023)
C10	Europe	102000		Agricultural	(Leitao et al., 2023)
C10	Europe	99000		Agricultural	(Leitao et al., 2023)
C11	Asia	2495		Agricultural	(Zhang et al., 2022)
C12	Asia	230		Agricultural	(Tajwar et al., 2022)
C14	Asia	8885		Agricultural	(Li et al., 2022b)
C15	Africa	161		Agricultural	(Chouchene et al., 2022)
C16	Europe	NA	100	Agricultural	(Muller et al., 2022)
C17	Asia	1220		Agricultural	(Mokhtarzadeh et al., 2022)
C18	Europe	8.88		Agricultural	(Weber et al., 2022)
C19	Asia	4107.4		Agricultural	(Tunali et al., 2022)
C2	Asia	3645		Agricultural	(Zhang et al., 2023a)
C20	Asia	504.5		Agricultural	(Nematollahi et al., 2022)
C21	Europe	94		Agricultural	(Colombini et al., 2022)
C22	Asia	899		Agricultural	(Li et al., 2022a)
C23	Asia	820		Agricultural	(Liu et al., 2023a)
C24	Asia	1675		Agricultural	(Lang et al., 2022)
C25	Asia	192.3333333		Agricultural	(Nguyen et al., 2022)
C26	Asia	247		Agricultural	(Park and Kim, 2022)
C27	Antarctica	NA		Agricultural	(Perfetti-Bolaño et al., 2022)
C28	Asia	7000		Agricultural	(Rezaei et al., 2022)
C29	Asia	NA		Agricultural	(Li et al., 2023d)
C3	Asia	69.7		Agricultural	(Feng et al., 2020)
C3	Asia	49.2		Agricultural	(Feng et al., 2020)
C3	Asia	75.7		Agricultural	(Feng et al., 2020)
C3	Asia	61.7		Agricultural	(Feng et al., 2020)

C30	South-America	540		Agricultural	(Corradini et al., 2021)
C30	South-America	420		Agricultural	(Corradini et al., 2021)
C30	South-America	2		Agricultural	(Corradini et al., 2021)
C30	South-America	1		Agricultural	(Corradini et al., 2021)
C31	Asia	NA		Agricultural	(Fakour et al., 2021)
C32	Africa	165.25		Agricultural	(Ragoobur et al., 2021)
C33	Asia	15090		Agricultural	(Tun et al., 2022)
C34	Europe	2118		Agricultural	(Dahl et al., 2021)
C35	Europe	182.4		Agricultural	(Schell et al., 2022)
C36	South-America	2000		Agricultural	(Álvarez-Lopeztello et al., 2021)
C36	South-America	3750		Agricultural	(Álvarez-Lopeztello et al., 2021)
C36	South-America	2000		Agricultural	(Álvarez-Lopeztello et al., 2021)
C36	South-America	2000		Agricultural	(Álvarez-Lopeztello et al., 2021)
C36	South-America	2000		Agricultural	(Álvarez-Lopeztello et al., 2021)
C37	Africa	438.15		Agricultural	(Boughattas et al., 2021)
C38	Asia	20		Agricultural	(Abbasi et al., 2021)
C39	Europe	2116		Agricultural	(Beriot et al., 2021)
C4	Europe	116620		Agricultural	(Lwanga et al., 2023)
C4	Europe	843808		Agricultural	(Lwanga et al., 2023)
C4	Europe	278845		Agricultural	(Lwanga et al., 2023)
C40	Asia	48.55		Agricultural	(Feng et al., 2021)
C41	Europe	3.7		Agricultural	(Harms et al., 2021)
C42	Asia	172.5		Agricultural	(Kim et al., 2021)
C42	Asia	1765		Agricultural	(Kim et al., 2021)
C43	Asia	4496		Agricultural	(Wang et al., 2021a)
C44	Asia	1444		Agricultural	(Yu et al., 2021)
C45	Asia	2420		Agricultural	(Ding et al. 2020)
C46	Asia	NA	0.04035	Agricultural	(Li et al., 2020e)
C47	Asia	3712.5		Agricultural	(Rafique et al., 2020)
C47	Asia	3770.833333		Agricultural	(Rafique et al., 2020)
C47	Asia	3429.166667		Agricultural	(Rafique et al., 2020)
C47	Asia	5775		Agricultural	(Rafique et al., 2020)
C47	Asia	4781.25		Agricultural	(Rafique et al., 2020)
C47	Asia	6250		Agricultural	(Rafique et al., 2020)

C47	Asia	3908.333333		Agricultural	(Rafique et al., 2020)
C47	Asia	4643.75		Agricultural	(Rafique et al., 2020)
C48	Asia	1132		Agricultural	(Duan et al., 2020)
C49	Asia	2020		Agricultural	(Chen et al., 2020a)
C5	North-America	42333.33333		Agricultural	(Koutnik et al., 2023)
C50	Asia	199.8333333		Agricultural	(Crossman et al., 2020)
C51	Asia	47.94		Agricultural	(Feng et al., 2020)
C52	Europe	1015.348039		Agricultural	(van den Berg et al., 2020)
C53	Asia	30.1		Agricultural	(Yang et al., 2021a)
C54	Asia	212.8333333		Agricultural	(Zhang et al., 2020a)
C55	Asia	503.3		Agricultural	(Zhou et al., 2020)
C56	Asia	111000		Agricultural	(Zhou et al., 2019)
C56	Asia	393000		Agricultural	(Zhou et al., 2019)
C56	Asia	331500		Agricultural	(Zhou et al., 2019)
C57	South-America	2040		Agricultural	(Corradini et al., 2019)
C58	Europe	NA	5	Agricultural	(Scheurer and Bigalke, 2018)
C59	Asia	70.25		Agricultural	(Liu et al., 2018)
C6	Asia	1169		Agricultural	(Su et al., 2023)
C6	Asia	1635.6		Agricultural	(Su et al., 2023)
C6	Asia	804.65		Agricultural	(Su et al., 2023)
C6	Asia	1504.216667		Agricultural	(Su et al., 2023)
C6	Asia	818.0857143		Agricultural	(Su et al., 2023)
C6	Asia	1131.695		Agricultural	(Su et al., 2023)
C6	Asia	666.1307692		Agricultural	(Su et al., 2023)
C60	Asia	18760		Agricultural	(Zhang and Liu, 2018)
C61	Europe	0.34		Agricultural	(Piehl et al., 2018)
C62	Australia	NA	7.764705882	Agricultural	(Fuller and Gautam, 2016)
C65	Asia	80.3		Agricultural	(Huang et al., 2020)
C65	Asia	308		Agricultural	(Huang et al., 2020)
C65	Asia	1075.6		Agricultural	(Huang et al., 2020)
C66	Asia	4300		Agricultural	(Liu et al., 2022a)
C66	Asia	2600		Agricultural	(Liu et al., 2022a)
C66	Asia	2600		Agricultural	(Liu et al., 2022a)
C66	Asia	2200		Agricultural	(Liu et al., 2022a)
C66	Asia	1800		Agricultural	(Liu et al., 2022a)
C66	Asia	1900		Agricultural	(Liu et al., 2022a)
C66	Asia	2200		Agricultural	(Liu et al., 2022a)
C66	Asia	2600		Agricultural	(Liu et al., 2022a)
C67	Asia	400		Agricultural	(Li et al., 2019b)
C67	Asia	800		Agricultural	(Li et al., 2019b)

C67	Asia	200		Agricultural	(Li et al., 2019b)
C67	Asia	500		Agricultural	(Li et al., 2019b)
C67	Asia	300		Agricultural	(Li et al., 2019b)
C67	Asia	400		Agricultural	(Li et al., 2019b)
C68	Europe	6000		Forest	(Cusworth et al., 2024)
C68	Europe	3333		Forest	(Cusworth et al., 2024)
C68	Europe	7333		Forest	(Cusworth et al., 2024)
C69	Europe	2610		Forest	(Cusworth et al., 2024)
C69	Europe	4770		Forest	(Cusworth et al., 2024)
C69	Europe	2720		Forest	(Cusworth et al., 2024)
C69	Europe	4610		Forest	(Cusworth et al., 2024)
C7	North-America	1533.333333		Forest	(Naderi Beni et al., 2023)
C71	North-America	NA	31100 n/m^2	Forest	(Helcoski et al., 2020)
C71	North-America	NA	26000 n/m^2	Grassland	(Helcoski et al., 2020)
C71	North-America	NA	9590 n/m^2	Grassland	(Helcoski et al., 2020)
C71	North-America	NA	25900 n/m^2	Grassland	(Helcoski et al., 2020)
C72	Europe	2302		Grassland	(van Schothorst et al., 2021)
C72	Europe	2179		Grassland	(van Schothorst et al., 2021)
C72	Europe	903		Grassland	(van Schothorst et al., 2021)
C72	Europe	848		Grassland	(van Schothorst et al., 2021)
C72	Europe	650		Grassland	(van Schothorst et al., 2021)
C72	Europe	1107		Greenhouse	(van Schothorst et al., 2021)
C73	Asia	5047		Greenhouse	(Yoon et al., 2024)
C73	Asia	1097		Greenhouse	(Yoon et al., 2024)
C73	Asia	3646		Greenhouse	(Yoon et al., 2024)
C73	Asia	2673		Greenhouse	(Yoon et al., 2024)
C73	Asia	4987		Greenhouse	(Yoon et al., 2024)
C74	Asia	6767		Greenhouse	(Wang et al., 2024)
C74	Asia	8507		Horticultural	(Wang et al., 2024)
C75	Europe	225		Horticultural	(Sa’adu and Farsang, 2022)
C75	Europe	75		Horticultural	(Sa’adu and Farsang, 2022)
C76	Asia	48		Horticultural	(Katsumi et al., 2021)
C77	Asia	NA	1.94	Horticultural	(Kumar and Sheela, 2021)
C77	Asia	NA	3.75	Horticultural	(Kumar and Sheela, 2021)
C77	Asia	NA	3.26	Horticultural	(Kumar and Sheela, 2021)
C77	Asia	NA	4.96	Horticultural	(Kumar and Sheela, 2021)
C78	Asia	1250		Horticultural	(Choi et al., 2020)
C78	Asia	250		Other	(Choi et al., 2020)

C78	Asia	600		Other	(Choi et al., 2020)
C78	Asia	750		Other	(Choi et al., 2020)
C78	Asia	160		Other	(Choi et al., 2020)
C78	Asia	1108		Other	(Choi et al., 2020)
C78	Asia	500		Other	(Choi et al., 2020)
C79	Europe	220		Other	(Isari et al., 2021)
C79	Europe	336		Other	(Isari et al., 2021)
C79	Europe	189		Other	(Isari et al., 2021)
C79	Europe	556		Other	(Isari et al., 2021)
C79	Europe	207		Other	(Isari et al., 2021)
C79	Europe	51		Other	(Isari et al., 2021)
C79	Europe	53		Plantation	(Isari et al., 2021)
C79	Europe	140		Plantation	(Isari et al., 2021)
C79	Europe	60		Plantation	(Isari et al., 2021)
C79	Europe	40		Plantation	(Isari et al., 2021)
C8	Asia	384		Urban	(Qiu et al., 2023)
C8	Asia	129		Urban	(Qiu et al., 2023)
C8	Asia	531		Urban	(Qiu et al., 2023)
C80	Africa	1000		Urban	(Kundu et al., 2022)
C80	Africa	750		Urban	(Kundu et al., 2022)
C80	Africa	780		Urban	(Kundu et al., 2022)
C80	Africa	1250		Urban	(Kundu et al., 2022)
C80	Africa	800		Urban	(Kundu et al., 2022)
C80	Africa	300		Urban	(Kundu et al., 2022)
C80	Africa	1000		Urban	(Kundu et al., 2022)
C80	Africa	250		Urban	(Kundu et al., 2022)
C80	Africa	900		Urban	(Kundu et al., 2022)
C80	Africa	100		Urban	(Kundu et al., 2022)
C81	North-America	117		Urban	(Adhikari et al., 2024)
C81	North-America	383		Urban	(Adhikari et al., 2024)
C81	North-America	500		Urban	(Adhikari et al., 2024)
C81	North-America	361		Urban	(Adhikari et al., 2024)
C82	Asia	2		Urban	(Himu et al., 2022)
C82	Asia	9		Urban	(Himu et al., 2022)
C82	Asia	6		Urban	(Himu et al., 2022)
C82	Asia	5		Urban	(Himu et al., 2022)
C83	South-America	870		Urban	(Huerta Lwanga et al., 2017b)
C84	Europe	NA	0.0003	Urban	(Ljung et al., 2018)

C84	Europe	NA	0.0034	Urban	(Ljung et al., 2018)
C84	Europe	NA	0.0003	Urban	(Ljung et al., 2018)
C85	Europe	12700000		Urban	(Meixner et al., 2020)
C86	Asia	163		Urban	(Zhang et al., 2020b)
C86	Asia	75		Urban	(Zhang et al., 2020b)
C87	Asia	185		Urban	(Bi et al., 2023)
C87	Asia	109		Urban	(Bi et al., 2023)
C87	Asia	148		Wetland	(Bi et al., 2023)
C9	Europe	296		Wetland	(Schöpfer et al., 2022)
MICROSOF	Europe	13		Wetland	(Palazot et al., 2024)
MICROSOF	Europe	32		Wetland	(Palazot et al., 2024)
MICROSOF	Europe	1.7		Wetland	(Palazot et al., 2024)
MICROSOF	Europe	23		Wetland	(Palazot et al., 2024)
Vollertsen and Hansen (2017)	Europe	71000		Wetland	(Vollertsen and Hansen, 2016)
Vollertsen and Hansen (2017)	Europe	145000		Wetland	(Vollertsen and Hansen, 2016)

REFERENCES

References

ABBASI, S., TURNER, A., HOSEINI, M. & AMIRI, H. 2021. Microplastics in the Lut and Kavir Deserts, Iran. *Environ Sci Technol,* 55**,** 5993-6000.

ADHIKARI, K., PEARCE, C. I., SANGUINET, K. A., BARY, A. I., CHOWDHURY, I., EGGLESTON, I., XING, B. & FLURY, M. 2024. Accumulation of microplastics in soil after long-term application of biosolids and atmospheric deposition. *Sci Total Environ,* 912**,** 168883.

ADMINISTRATOR OF THE GOVERNMENT OF CANADA IN COUNCIL 2021. Order Adding a Toxic Substance to Schedule 1 to the Canadian Environmental Protection Act, 1999: SOR/2021-86.

AKDOGAN, Z. & GUVEN, B. 2019. Microplastics in the environment: A critical review of current understanding and identification of future research needs. *Environmental pollution,* 254.

ÁLVAREZ-LOPEZTELLO, J., ROBLES, C. & DEL CASTILLO, R. F. 2021. Microplastic pollution in neotropical rainforest, savanna, pine plantations, and pasture soils in lowland areas of Oaxaca, Mexico: Preliminary results. *Ecological Indicators,* 121.

AMOBONYE, A., BHAGWAT, P., RAVEENDRAN, S., SINGH, S. & PILLAI, S. 2021. Environmental Impacts of Microplastics and Nanoplastics: A Current Overview - PubMed. *Frontiers in microbiology,* 12.

ANDRADY, A. L. 2011. Microplastics in the marine environment. *Mar Pollut Bull,* 62**,** 1596-605.

ARAUJO, C. F., NOLASCO, M. M., RIBEIRO, A. M. P. & RIBEIRO-CLARO, P. J. A. 2018. Identification of microplastics using Raman spectroscopy: Latest developments and future prospects. *Water Res,* 142**,** 426-440.

ATEIA, M., ERSAN, G., ALALM, M. G., BOFFITO, D. C. & KARANFIL, T. 2022. Emerging investigator series: microplastic sources, fate, toxicity, detection, and interactions with micropollutants in aquatic ecosystems - a review of reviews. *Environ Sci Process Impacts,* 24**,** 172-195.

AUSTEN, K., MACLEAN, J., BALANZATEGUI, D. & HOLKER, F. 2022. Microplastic inclusion in birch tree roots. *Sci Total Environ,* 808**,** 152085.

BARBOSA, A. E. & FERNANDES, J. N. 2021. Review of tools for road runoff quality prediction and application to European roads. *Water Sci Technol,* 84**,** 2228-2241.

BEGHETTO, V., SOLE, R., BURANELLO, C., AL-ABKAL, M. & FACCHIN, M. 2021. Recent Advancements in Plastic Packaging Recycling: A Mini-Review. *Materials,* 14.

BERIOT, N., PEEK, J., ZORNOZA, R., GEISSEN, V. & HUERTA LWANGA, E. 2021. Low density-microplastics detected in sheep faeces and soil: A case study from the intensive vegetable farming in Southeast Spain. *Sci Total Environ,* 755**,** 142653.

BERIOT, N., ZORNOZA, R., LWANGA, E. H., ZOMER, P., VAN SCHOTHORST, B., OZBOLAT, O., LLORET, E., ORTEGA, R., MIRALLES, I., HARKES, P., VAN STEENBRUGGE, J. & GEISSEN, V. 2023. Intensive vegetable production under plastic mulch: A field study on soil plastic and pesticide residues and their effects on the soil microbiome. *Sci Total Environ,* 900**,** 165179.

BERSET, F. C. D., STOLL, S., BERSET, F. C. D. & STOLL, S. 2024. Microplastic Contamination in Field-Side Composting in Geneva, Switzerland (CH). *Microplastics,* 3**,** 477-491.

BERTINI, S. C. B. & AZEVEDO, L. C. B. 2022. Soil microbe contributions in the regulation of the global carbon cycle. *Microbiome Under Changing Climate.*

BI, D., WANG, B., LI, Z., ZHANG, Y., KE, X., HUANG, C., LIU, W., LUO, Y., CHRISTIE, P. & WU, L. 2023. Occurrence and distribution of microplastics in coastal plain soils under three land-use types. *Sci Total Environ,* 855**,** 159023.

BLÄSING, M. & AMELUNG, W. 2018. Plastics in soil: Analytical methods and possible sources. *Sci Total Environ,* 612**,** 422-435.

BOOTS, B., RUSSELL, C. W. & GREEN, D. S. 2019. Effects of Microplastics in Soil Ecosystems: Above and Below Ground. *Environ Sci Technol,* 53**,** 11496-11506.

BOSKER, T., BOUWMAN, L. J., BRUN, N. R., BEHRENS, P. & VIJVER, M. G. 2019. Microplastics accumulate on pores in seed capsule and delay germination and root growth of the terrestrial vascular plant Lepidium sativum. *Chemosphere,* 226**,** 774-781.

BOUCHER, J. & FRIOT, D. 2017. Primary Microplastics in the Oceans:
a Global Evaluation of Sources.

BOUGHATTAS, I., HATTAB, S., ZITOUNI, N., MKHININI, M., MISSAWI, O., BOUSSERRHINE, N. & BANNI, M. 2021. Assessing the presence of microplastic particles in Tunisian agriculture soils and their potential toxicity effects using Eisenia andrei as bioindicator. *Sci Total Environ,* 796**,** 148959.

BOUWSTRA, J., PILGRAM, G., GOORIS, G., KOERTEN, H. & PONEC, M. 2001. New aspects of the skin barrier organization. *Skin pharmacology and applied skin physiology,* 14 Suppl 1.

BRIDSON, J. H., GAUGLER, E. C., SMITH, D. A., NORTHCOTT, G. L. & GAW, S. 2021. Leaching and extraction of additives from plastic pollution to inform environmental risk: A multidisciplinary review of analytical approaches. *J Hazard Mater,* 414**,** 125571.

BRONICK, C. J. & LAL, R. 2005. Soil structure and management: a review. *Geoderma,* 124**,** 3-22.

BROUWER, M. T., CHACON, F. A. & VAN VELZEN, E. U. T. 2020. Effect of recycled content and rPET quality on the properties of PET bottles, part I: Optical and mechanical properties. *Packaging Technology and Science,* 33.

BROWN, E., MACDONALD, A., ALLEN, S. & ALLEN, D. 2023a. The potential for a plastic recycling facility to release microplastic pollution and possible filtration remediation effectiveness. *Journal of Hazardous Materials Advances,* 10.

BROWN, R. W., CHADWICK, D. R., ZANG, H., GRAF, M., LIU, X., WANG, K., GREENFIELD, L. M. & JONES, D. L. 2023b. Bioplastic (PHBV) addition to soil alters microbial community structure and negatively affects plant-microbial metabolic functioning in maize. *Journal of Hazardous Materials,* 441.

BROWNE, M. A., CRUMP, P., NIVEN, S. J., TEUTEN, E., TONKIN, A., GALLOWAY, T. & THOMPSON, R. 2011. Accumulation of microplastic on shorelines woldwide: sources and sinks. *Environ Sci Technol,* 45**,** 9175-9.

BUCCI, K., ROCHMAN, C. M., BUCCI, K. & ROCHMAN, C. M. 2022. Microplastics: a multidimensional contaminant requires a multidimensional framework for assessing risk. *Microplastics and Nanoplastics 2022 2:1,* 2.

BÜKS, F. & KAUPENJOHANN, M. 2020. Global concentrations of microplastics in soils – a review. *Soil,* 6**,** 649-662.

CARNERO, A. R., LESTIDO-CARDAMA, A., LOUREIRO, P. V., BARBOSA-PEREIRA, L., QUIRÓS, A. R. B. D., SENDÓN, R., RAMÍREZ CARNERO, A., LESTIDO-CARDAMA, A., VAZQUEZ LOUREIRO, P., BARBOSA-PEREIRA, L., RODRÍGUEZ BERNALDO DE QUIRÓS, A. & SENDÓN, R. 2021. Presence of Perfluoroalkyl and Polyfluoroalkyl Substances (PFAS) in Food Contact Materials (FCM) and Its Migration to Food. *Foods 2021, Vol. 10, Page 1443,* 10.

CARR, S. A., LIU, J. & TESORO, A. G. 2016. Transport and fate of microplastic particles in wastewater treatment plants. *Water Res,* 91**,** 174-82.

CARTER, M. R. & GREGORICH, E. G. 2007. Soil Sampling and Methods of Analysis.

CASTAN, S., HENKEL, C., HÜFFER, T. & HOFMANN, T. 2021. Microplastics and nanoplastics barely enhance contaminant mobility in agricultural soils. *Communications Earth & Environment,* 2.

CASTORINA, R., BRADMAN, A., STAPLETON, H. M., BUTT, C., AVERY, D., HARLEY, K. G., GUNIER, R. B., HOLLAND, N. & ESKENAZI, B. 2017. Current-use flame retardants: Maternal exposure and neurodevelopment in children of the CHAMACOS cohort. *Chemosphere,* 189.

CHEMLINKED. 2019. *Thailand to ban microbeads in cosmetics* [Online]. Available: https://cosmetic.chemlinked.com/news/cosmetic-news/thailand-ban-microbeads-cosmetics [Accessed].

CHEN, Y., LENG, Y., LIU, X. & WANG, J. 2020a. Microplastic pollution in vegetable farmlands of suburb Wuhan, central China. *Environmental Pollution,* 257.

CHEN, Y., WEN, D., PEI, J., FEI, Y., OUYANG, D., ZHANG, H. & LUO, Y. 2020b. Identification and quantification of microplastics using Fourier-transform infrared spectroscopy: Current status and future prospects. *Current Opinion in Environmental Science & Health,* 18**,** 14-19.
CHENG, S., JESSICA, YOSHIKAWA, K. & CROSS, J. S. 2023. Effects of nano/microplastics on the growth and reproduction of the microalgae, bacteria, fungi, and Daphnia magna in the microcosms. *Environmental Technology & Innovation,* 31.
CHESHIRE, A., E, A., BARBIÈRE, J., COHEN, Y., EVANS, S., JARAYABHAND, S., JEFTIC, L., JUNG, R.-T., KINSEY, S., KUSUI, E. T., LAVINE, I., MANYARA, P., OOSTERBAAN, L., PEREIRA, M. A., SHEAVLY, S., TKALIN, A., VARADARAJAN, S., WENNEKER, B. & WESTPHALEN, G. 2009. UNEP/IOC guidelines on survey monitoring of marine litter. Regional seas reports and studies No. 186 IOC Technical Series No. 83.
CHOI, Y. R., KIM, Y.-N., YOON, J.-H., DICKINSON, N. & KIM, K.-H. 2020. Plastic contamination of forest, urban, and agricultural soils: a case study of Yeoju City in the Republic of Korea. *Journal of Soils and Sediments,* 21**,** 1962-1973.
CHOUCHENE, K., NACCI, T., MODUGNO, F., CASTELVETRO, V. & KSIBI, M. 2022. Soil contamination by microplastics in relation to local agricultural development as revealed by FTIR, ICP-MS and pyrolysis-GC/MS. *Environ Pollut,* 303**,** 119016.
COLE, M., LINDEQUE, P., HALSBAND, C. & GALLOWAY, T. S. 2011. Microplastics as contaminants in the marine environment: a review. *Mar Pollut Bull,* 62**,** 2588-97.
COLOMBINI, G., RUMPEL, C., HOUOT, S., BIRON, P. & DIGNAC, M. F. 2022. A long-term field experiment confirms the necessity of improving biowaste sorting to decrease coarse microplastic inputs in compost amended soils. *Environ Pollut,* 315**,** 120369.
CONTI, G. O., FERRANTE, M., BANNI, M., FAVARA, C., NICOLOSI, I., CRISTALDI, A., FIORE, M. & ZUCCARELLO, P. 2020. Micro- and nano-plastics in edible fruit and vegetables. The first diet risks assessment for the general population. *Environmental Research,* 187.
COOK, E., DERKS, M. & VELIS, C. A. 2023. Plastic waste reprocessing for circular economy: A systematic scoping review of risks to occupational and public health from legacy substances and extrusion. *Science of The Total Environment,* 859.
COOPER, D. A. & CORCORAN, P. L. 2010. Effects of mechanical and chemical processes on the degradation of plastic beach debris on the island of Kauai, Hawaii. *Mar Pollut Bull,* 60**,** 650-4.
CORRADINI, F., CASADO, F., LEIVA, V., HUERTA-LWANGA, E. & GEISSEN, V. 2021. Microplastics occurrence and frequency in soils under different land uses on a regional scale. *Sci Total Environ,* 752**,** 141917.
CORRADINI, F., MEZA, P., EGUILUZ, R., CASADO, F., HUERTA-LWANGA, E. & GEISSEN, V. 2019. Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. *Sci Total Environ,* 671**,** 411-420.
COSTA, J. P. D., AVELLAN, A., MOUNEYRAC, C., DUARTE, A. & ROCHA-SANTOS, T. 2023. Plastic additives and microplastics as emerging contaminants: Mechanisms and analytical assessment. *TrAC Trends in Analytical Chemistry,* 158.
COURTNEY, A., BAKER, J. & BAMFORD, H. 2008. Proceedings of the international research workshop on the occurrence, effects, and fate of microplastic marine debris. NOAA Marine debris devision.
COVERNTON, G. A., PEARCE, C. M., GURNEY-SMITH, H. J., CHASTAIN, S. G., ROSS, P. S., DOWER, J. F. & DUDAS, S. E. 2019. Size and shape matter: A preliminary analysis of microplastic sampling technique in seawater studies with implications for ecological risk assessment. *Sci Total Environ,* 667**,** 124-132.
CRABTREE, B., MOY, F., WHITEHEAD, M. & ROE, A. 2006. Monitoring pollutants in highway runoff. *Water and Environment Journal,* 20**,** 295-295.
CROSSMAN, J., HURLEY, R. R., FUTTER, M. & NIZZETTO, L. 2020. Transfer and transport of microplastics from biosolids to agricultural soils and the wider environment. *Sci Total Environ,* 724**,** 138334.
CUSWORTH, S. J., DAVIES, W. J., MCAINSH, M. R., GREGORY, A. S., STORKEY, J. & STEVENS, C. J. 2024. Agricultural fertilisers contribute substantially to microplastic concentrations in UK soils. *Communications Earth & Environment,* 5.
CVERENKÁROVÁ, K., VALACHOVIČOVÁ, M., MACKUĽAK, T., ŽEMLIČKA, L. & BÍROŠOVÁ, L. 2021. Microplastics in the Food Chain. *Life,* 11.

DA COSTA, J. P., SANTOS, P. S. M., DUARTE, A. C. & ROCHA-SANTOS, T. 2016. (Nano)plastics in the environment - Sources, fates and effects. *Sci Total Environ,* 566-567**,** 15-26.
DAHL, M., BERGMAN, S., BJORK, M., DIAZ-ALMELA, E., GRANBERG, M., GULLSTROM, M., LEIVA-DUENAS, C., MAGNUSSON, K., MARCO-MENDEZ, C., PINEIRO-JUNCAL, N. & MATEO, M. A. 2021. A temporal record of microplastic pollution in Mediterranean seagrass soils. *Environ Pollut,* 273**,** 116451.
DE SOUZA MACHADO, A. A., KLOAS, W., ZARFL, C., HEMPEL, S. & RILLIG, M. C. 2018a. Microplastics as an emerging threat to terrestrial ecosystems. *Glob Chang Biol,* 24**,** 1405-1416.
DE SOUZA MACHADO, A. A., LAU, C. W., KLOAS, W., BERGMANN, J., BACHELIER, J. B., FALTIN, E., BECKER, R., GORLICH, A. S. & RILLIG, M. C. 2019. Microplastics Can Change Soil Properties and Affect Plant Performance. *Environ Sci Technol,* 53**,** 6044-6052.
DE SOUZA MACHADO, A. A., LAU, C. W., TILL, J., KLOAS, W., LEHMANN, A., BECKER, R. & RILLIG, M. C. 2018b. Impacts of microplastics on the soil biophysical environment. *Environ Sci Technol,* 52**,** 9656-9665.
DEY, T. 2022. Microplastic pollutant detection by Surface Enhanced Raman Spectroscopy (SERS): a mini-review. *Nanotechnology for Environmental Engineering,* 8**,** 41-48.
DOHERTY, B., HAMMEL, S., DANIELS, J., STAPLETON, H. & HOFFMAN, K. 2019. Organophosphate Esters: Are These Flame Retardants and Plasticizers Affecting Children's Health? *Current environmental health reports,* 6.
DONG, Y., GAO, M., QIU, W. & SONG, Z. 2021. Uptake of microplastics by carrots in presence of As (III): Combined toxic effects. *Journal of Hazardous Materials,* 411.
DUAN, Z., ZHAO, S., ZHAO, L., DUAN, X., XIE, S., ZHANG, H., LIU, Y., PENG, Y., LIU, C. & WANG, L. 2020. Microplastics in Yellow River Delta wetland: Occurrence, characteristics, human influences, and marker. *Environ Pollut,* 258**,** 113232.
DUIKER, S. W., RHOTON, F. E., TORRENT, J., SMECK, N. E. & LAL, R. 2003. Iron (Hydr)Oxide Crystallinity Effects on Soil Aggregation. *Soil Science Society of America Journal,* 67.
DUIS, K. & COORS, A. 2016. Microplastics in the aquatic and terrestrial environment: sources (with a specific focus on personal care products), fate and effects. *Environ Sci Eur,* 28**,** 2.
DUMICHEN, E., EISENTRAUT, P., BANNICK, C. G., BARTHEL, A. K., SENZ, R. & BRAUN, U. 2017. Fast identification of microplastics in complex environmental samples by a thermal degradation method. *Chemosphere,* 174**,** 572-584.
ECHA 2019. ANNEX XV Restriction report proposal for restriction: intentionally added microplastics. Helsinki, Finland: European Chemical Agency.
ECHA. 2025. *Microplastics* [Online]. Available: https://echa.europa.eu/hot-topics/microplastics [Accessed].
EFSA. 25th EFSA colloquium: A coordinated approach to assess the human health risks of micro- and nanoplastics in food. EFSA Scientific Colloquium 25, May 6 and 7, 2021 2021. European Food Safety Authority (EFSA).
EMERSON, W. W. & GREENLAND, D. J. 1990. Soil aggregates - Formation and stability. *Soil colloids and their associations in aggregates.* Boston, MA: Springer US.
EN-NEJMY, K., EL HAYANY, B., AL-ALAWI, M., JEMO, M., HAFIDI, M. & EL FELS, L. 2024. Microplastics in soil: A comprehensive review of occurrence, sources, fate, analytical techniques and potential impacts. *Ecotoxicol Environ Saf,* 288**,** 117332.
EPA. 2024. *Microbeads ban – is your product affected?* [Online]. Available: https://www.epa.govt.nz/news-and-alerts/alerts/microbeads-ban-is-your-product-affected [Accessed 28/02/2025].
EPPEHIMER, D. E., HAMDHANI, H., HOLLIEN, K. D., NEMEC, Z. C., LEE, L. N., QUANRUD, D. M. & BOGAN, M. T. 2021. Impacts of baseflow and flooding on microplastic pollution in an effluent-dependent arid land river in the USA. *Environ Sci Pollut Res Int,* 28**,** 45375-45389.
EUROPEAN COMMISSION. 2021. *Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions: Pathway to a Healthy Planet for All EU Action Plan: 'Towards Zero Pollution for Air, Water and Soil* [Online]. Available: https://eur-lex.europa.eu/legal-content/EN/TXT/HTML/?uri=CELEX:52021DC0400 [Accessed].
EUROPEAN COMMISSION. 2023. *Microplastics* [Online]. Available: https://environment.ec.europa.eu/topics/plastics/microplastics_en [Accessed 31/01/2025].

EUROPEANBIOPLASTICS. 2024. *What are bioplastics?* [Online]. Available: https://www.european-bioplastics.org/bioplastics/ [Accessed 28/02/2025].
EVANGELIOU, N., GRYTHE, H., KLIMONT, Z., HEYES, C., ECKHARDT, S., LOPEZ-APARICIO, S., STOHL, A., EVANGELIOU, N., GRYTHE, H., KLIMONT, Z., HEYES, C., ECKHARDT, S., LOPEZ-APARICIO, S. & STOHL, A. 2020. Atmospheric transport is a major pathway of microplastics to remote regions. *Nature Communications,* 11.
EVERAERT, G., VAN CAUWENBERGHE, L., DE RIJCKE, M., KOELMANS, A., MEES, J., VANDEGEHUCHTE, M. & JANSSEN, C. 2018. Risk assessment of microplastics in the ocean: Modelling approach and first conclusions. *Environmental pollution,* 242.
FAKOUR, H., LO, S.-L., YOASHI, N. T., MASSAO, A. M., LEMA, N. N., MKHONTFO, F. B., JOMALEMA, P. C., JUMANNE, N. S., MBUYA, B. H., MTWEVE, J. T. & IMANI, M. 2021. Quantification and Analysis of Microplastics in Farmland Soils: Characterization, Sources, and Pathways. *Agriculture,* 11.
FEI, Y., HUANG, S., ZHANG, H., TONG, Y., WEN, D., XIA, X., WANG, H., LUO, Y. & BARCELO, D. 2020. Response of soil enzyme activities and bacterial communities to the accumulation of microplastics in an acid cropped soil. *Sci Total Environ,* 707**,** 135634.
FEITEIRO, J., MARIANA, M. & CAIRRÃO, E. 2021. Health toxicity effects of brominated flame retardants: From environmental to human exposure. *Environmental Pollution,* 285.
FELSING, S., KOCHLEUS, C., BUCHINGER, S., BRENNHOLT, N., STOCK, F. & REIFFERSCHEID, G. 2018. A new approach in separating microplastics from environmental samples based on their electrostatic behavior. *Environ Pollut,* 234**,** 20-28.
FENG, S., LU, H. & LIU, Y. 2021. The occurrence of microplastics in farmland and grassland soils in the Qinghai-Tibet plateau: Different land use and mulching time in facility agriculture. *Environ Pollut,* 279**,** 116939.
FENG, S., LU, H., TIAN, P., XUE, Y., LU, J., TANG, M. & FENG, W. 2020. Analysis of microplastics in a remote region of the Tibetan Plateau: Implications for natural environmental response to human activities. *Sci Total Environ,* 739**,** 140087.
FENG, Y., TU, C., LI, R., WU, D., YANG, J., XIA, Y., PEIJNENBURG, W. & LUO, Y. 2023. A systematic review of the impacts of exposure to micro- and nano-plastics on human tissue accumulation and health. *Eco Environ Health,* 2**,** 195-207.
FITRI, D. F., ARMIN, F. & DJAMAAN, A. 2021. Use of polymers as coatings for slow release fertilizers. *IOSR Journal of pharmacy and biological sciences,* 16**,** 22-31.
FOEKEMA, E. M., DE GRUIJTER, C., MERGIA, M. T., VAN FRANEKER, J. A., MURK, A. J. & KOELMANS, A. A. 2013. Plastic in north sea fish. *Environ Sci Technol,* 47**,** 8818-24.
FRIAS, J. & NASH, R. 2019. Microplastics: Finding a consensus on the definition. *Mar Pollut Bull,* 138**,** 145-147.
FULLER, S. & GAUTAM, A. 2016. A Procedure for Measuring Microplastics using Pressurized Fluid Extraction. *Environ Sci Technol,* 50**,** 5774-80.
GALGANI, F., HANKE, G., WERNER, S. & DE VREES, L. 2013. Marine litter within the European Marine Strategy Framework Directive. *ICES Journal of Marine Science,* 70**,** 1055-1064.
GAO, B., SAIERS, J. E. & RYAN, J. 2006. Pore-scale mechanisms of colloid deposition and mobilization during steady and transient flow through unsaturated granular media. *Water Resources Research,* 42.
GAO, M., LIU, Y., DONG, Y. & SONG, Z. 2021. Effect of polyethylene particles on dibutyl phthalate toxicity in lettuce (Lactuca sativa L.). *J Hazard Mater,* 401**,** 123422.
GARCIA, J. M. & ROBERTSON, M. L. 2017. The future of plastics recycling. *Science,* 358.
GERBEN, G. 2022. Methods for calculating the emissions of transport in the Netherlands.
GEYER, R., JAMBECK, J. R. & LAW, K. L. 2017. Production, use, and fate of all plastics ever made. *Science advances,* 3.
GIGAULT, J., HALLE, A. T., BAUDRIMONT, M., PASCAL, P. Y., GAUFFRE, F., PHI, T. L., EL HADRI, H., GRASSL, B. & REYNAUD, S. 2018. Current opinion: What is a nanoplastic? *Environ Pollut,* 235**,** 1030-1034.
GIONFRA, S. 2018. Plastic pollution in soil. Institute for European Environmental Policy (IEEP).
GOERTZ, H. M. 2000. Controlled Release Technology, Agricultural. *Kirk-Othmer Encyclopedia of Chemical Technology*.

GONG, W., ZHANG, W., JIANG, M., LI, S., LIANG, G., BU, Q., XU, L., ZHU, H. & LU, A. 2021. Species-dependent response of food crops to polystyrene nanoplastics and microplastics. *Sci Total Environ,* 796**,** 148750.
GOPINATH, P., PARVATHI, V., YOGHALAKSHMI, N., KUMAR, S., ATHULYA, P., MUKHERJEE, A. & CHANDRASEKARAN, N. 2022. Plastic particles in medicine: A systematic review of exposure and effects to human health. *Chemosphere,* 303.
GOUIN, T., BECKER, R. A., COLLOT, A.-G., DAVIS, J. W., HOWARD, B., INAWAKA, K., LAMPI, M., RAMON, B. S., SHI, J. & HOPP, P. W. 2019. Toward the Development and Application of an Environmental Risk Assessment Framework for Microplastic. *Environmental Toxicology and Chemistry,* 38.
GUI, J., SUN, Y., WANG, J., CHEN, X., ZHANG, S. & WU, D. 2021. Microplastics in composting of rural domestic waste: abundance, characteristics, and release from the surface of macroplastics. *Environmental Pollution,* 274.
GUO, M., NOORI, R. & ABOLFATHI, S. 2024a. Microplastics in freshwater systems: Dynamic behaviour and transport processes. *Resources, Conservation and Recycling,* 205.
GUO, W., YE, Z., ZHAO, Y., LU, Q., SHEN, B., ZHANG, X., ZHANG, W., CHEN, S. & LI, Y. 2024b. Effects of different microplastic types on soil physicochemical properties, enzyme activities, and bacterial communities. *Ecotoxicol Environ Saf,* 286**,** 117219.
GUO, Z., LI, P., YANG, X., WANG, Z., LU, B., CHEN, W., WU, Y., LI, G., ZHAO, Z., LIU, G., RITSEMA, C., GEISSEN, V. & XUE, S. 2022. Soil texture is an important factor determining how microplastics affect soil hydraulic characteristics. *Environment International,* 165.
HALE, S. E., ARP, H. P. H., SCHLIEBNER, I. & NEUMANN, M. 2020. Persistent, mobile and toxic (PMT) and very persistent and very mobile (vPvM) substances pose an equivalent level of concern to persistent, bioaccumulative and toxic (PBT) and very persistent and very bioaccumulative (vPvB) substances under REACH. *Environmental Sciences Europe,* 32.
HARMS, I. K., DIEKOTTER, T., TROEGEL, S. & LENZ, M. 2021. Amount, distribution and composition of large microplastics in typical agricultural soils in Northern Germany. *Sci Total Environ,* 758**,** 143615.
HARTMANN, N. B., HUFFER, T., THOMPSON, R. C., HASSELLOV, M., VERSCHOOR, A., DAUGAARD, A. E., RIST, S., KARLSSON, T., BRENNHOLT, N., COLE, M., HERRLING, M. P., HESS, M. C., IVLEVA, N. P., LUSHER, A. L. & WAGNER, M. 2019. Are We Speaking the Same Language? Recommendations for a Definition and Categorization Framework for Plastic Debris. *Environ Sci Technol,* 53**,** 1039-1047.
HASSAN, F., PRASETYA, K. D., HANUN, J. N., BUI, H. M., RAJENDRAN, S., KATARIA, N., KHOO, K. S., WANG, Y.-F., YOU, S.-J. & JIANG, J.-J. 2023. Microplastic contamination in sewage sludge: Abundance, characteristics, and impacts on the environment and human health. *Environmental Technology & Innovation,* 31.
HATALEY, E. K., MCILWRAITH, H. K., ROY, D. & ROCHMAN, C. M. 2023. Towards a management strategy for microplastic pollution in the Laurentian Great Lakes—ecological risk assessment and management (part 2). *Canadian Journal of Fisheries and Aquatic Sciences,* 80**,** 1669-1678.
HE, B., LIU, Z., WANG, X., LI, M., LIN, X., XIAO, Q. & HU, J. 2024. Dosage and exposure time effects of two micro(nono)plastics on arbuscular mycorrhizal fungal diversity in two farmland soils planted with pepper (Capsicum annuum L.). *Sci Total Environ,* 917**,** 170216.
HEINZE, W. M., MITRANO, D. M., LAHIVE, E., KOESTEL, J. & CORNELIS, G. 2021. Nanoplastic Transport in Soil via Bioturbation by Lumbricus terrestris. *Environ Sci Technol,* 55**,** 16423-16433.
HELCOSKI, R., YONKOS, L. T., SANCHEZ, A. & BALDWIN, A. H. 2020. Wetland soil microplastics are negatively related to vegetation cover and stem density. *Environ Pollut,* 256**,** 113391.
HENKEL, C., HÜFFER, T. & HOFMANN, T. 2022. Polyvinyl Chloride Microplastics Leach Phthalates into the Aquatic Environment over Decades. *Environmental Science & Technology,* 56.
HERMABESSIERE, L., HIMBER, C., BORICAUD, B., KAZOUR, M., AMARA, R., CASSONE, A. L., LAURENTIE, M., PAUL-PONT, I., SOUDANT, P., DEHAUT, A. & DUFLOS, G. 2018. Optimization, performance, and application of a pyrolysis-GC/MS method for the identification of microplastics. *Anal Bioanal Chem,* 410**,** 6663-6676.
HERNANDEZ, L. M., XU, E. G., LARSSON, H. C. E., TAHARA, R., MAISURIA, V. B. & TUFENKJI, N. 2019. Plastic Teabags Release Billions of Microparticles and Nanoparticles into Tea. *Environmental Science & Technology,* 53.

HEUCHAN, S. M., FAN, B., KOWALSKI, J. J., GILLIES, E. R. & HENRY, H. A. L. 2019. Development of Fertilizer Coatings from Polyglyoxylate-Polyester Blends Responsive to Root-Driven pH Change. *J Agric Food Chem,* 67**,** 12720-12729.
HIMU, M. M., AFRIN, S., AKBOR, M. A., BAKAR SIDDIQUE, M. A., UDDIN, M. K. & RAHMAN, M. M. 2022. Assessment of microplastics contamination on agricultural farmlands in central Bangladesh. *Case Studies in Chemical and Environmental Engineering,* 5.
HOANG, V.-H., NGUYEN, M.-K., HOANG, T.-D., RANGEL-BUITRAGO, N., LIN, C., PHAM, M.-T., HA, M. C., NGUYEN, T. P., SHAABAN, M., CHANG, S. W. & NGUYEN, D. D. 2025. Microplastic characteristics, transport, risks, and remediation in groundwater: a review. *Environmental Chemistry Letters*.
HOEKE, S., VAN WIJNEN, J., KRIKKE, H., LÖHR, A. & RAGAS, A. M. J. 2024. Mapping the tire supply chain and its microplastics emissions using a multi-stakeholder approach. *Resources, Conservation and Recycling,* 203.
HOFMANN, T., GHOSHAL, S., TUFENKJI, N., ADAMOWSKI, J. F., BAYEN, S., CHEN, Q., DEMOKRITOU, P., FLURY, M., HÜFFER, T., IVLEVA, N. P., JI, R., LEASK, R. L., MARIC, M., MITRANO, D. M., SANDER, M., PAHL, S., RILLIG, M. C., WALKER, T. R., WHITE, J. C. & WILKINSON, K. J. 2023. Plastics can be used more sustainably in agriculture. *Communications Earth & Environment,* 4.
HONG, M. & CHEN, E. Y. X. 2017. Chemically recyclable polymers: a circular economy approach to sustainability. *Green Chemistry,* 19**,** 3692-3706.
HORTON, A. A., WALTON, A., SPURGEON, D. J., LAHIVE, E. & SVENDSEN, C. 2017. Microplastics in freshwater and terrestrial environments: Evaluating the current understanding to identify the knowledge gaps and future research priorities. *Sci Total Environ,* 586**,** 127-141.
HU, C., LU, B., GUO, W., TANG, X., WANG, X., XUE, Y., WANG, L. & HE, X. 2021. Distribution of microplastics in mulched soil in Xinjiang, China. *International Journal of Agricultural and Biological Engineering,* 14**,** 196-204.
HU, J., HE, D., ZHANG, X., LI, X., CHEN, Y., WEI, G., ZHANG, Y., OK, Y. S. & LUO, Y. 2022. National-scale distribution of micro(meso)plastics in farmland soils across China: Implications for environmental impacts. *J Hazard Mater,* 424**,** 127283.
HU, Z.-Y. & JIANG, H. 2024. Vital Threshold and Underlying Mechanism for the Complete Remediation of Oil and Microplastic Co-Contaminated Soil by Fast Pyrolysis. *ACS ES&T Engineering,* 4**,** 2198-2208.
HUANG, D., WANG, X., YIN, L., CHEN, S., TAO, J., ZHOU, W., CHEN, H., ZHANG, G. & XIAO, R. 2022. Research progress of microplastics in soil-plant system: Ecological effects and potential risks. *Sci Total Environ,* 812**,** 151487.
HUANG, J., CHEN, H., ZHENG, Y., YANG, Y., ZHANG, Y. & GAO, B. 2021. Microplastic pollution in soils and groundwater: Characteristics, analytical methods and impacts. *Chemical Engineering Journal,* 425.
HUANG, Y., LIU, Q., JIA, W., YAN, C. & WANG, J. 2020. Agricultural plastic mulching as a source of microplastics in the terrestrial environment. *Environ Pollut,* 260**,** 114096.
HUERTA LWANGA, E., GERTSEN, H., GOOREN, H., PETERS, P., SALANKI, T., VAN DER PLOEG, M., BESSELING, E., KOELMANS, A. A. & GEISSEN, V. 2016. Microplastics in the Terrestrial Ecosystem: Implications for Lumbricus terrestris (Oligochaeta, Lumbricidae). *Environ Sci Technol,* 50**,** 2685-91.
HUERTA LWANGA, E., GERTSEN, H., GOOREN, H., PETERS, P., SALANKI, T., VAN DER PLOEG, M., BESSELING, E., KOELMANS, A. A. & GEISSEN, V. 2017a. Incorporation of microplastics from litter into burrows of Lumbricus terrestris. *Environ Pollut,* 220**,** 523-531.
HUERTA LWANGA, E., MENDOZA VEGA, J., KU QUEJ, V., CHI, J. L. A., SANCHEZ DEL CID, L., CHI, C., ESCALONA SEGURA, G., GERTSEN, H., SALANKI, T., VAN DER PLOEG, M., KOELMANS, A. A. & GEISSEN, V. 2017b. Field evidence for transfer of plastic debris along a terrestrial food chain. *Sci Rep,* 7**,** 14071.
HURLEY, R. R. & NIZZETTO, L. 2018. Fate and occurrence of micro(nano)plastics in soils: Knowledge gaps and possible risks. *Current Opinion in Environmental Science & Health,* 1**,** 6-11.
IGNATYEV, I. A., THIELEMANS, W. & BEKE, B. V. 2014. Recycling of Polymers: A Review. *ChemSusChem,* 7.
IÑIGUEZ, M. E., CONESA, J. A., FULLANA, A., IÑIGUEZ, M. E., CONESA, J. A. & FULLANA, A. 2017. Microplastics in Spanish Table Salt. *Scientific Reports,* 7.

INTERREG BALTIC SEA REGION. 2024. *How plastic additives affect your health* [Online]. Available: https://interreg-baltic.eu/project-posts/baltiplast/additives-in-plastics/ [Accessed 02/02/2025].
ISARI, E. A., PAPAIOANNOU, D., KALAVROUZIOTIS, I. K. & KARAPANAGIOTI, H. K. 2021. Microplastics in Agricultural Soils: A Case Study in Cultivation of Watermelons and Canning Tomatoes. *Water,* 13.
ISWAHYUDI, I., SUTANTO, A., WIDODO, W., WARKOYO, W., GARFANSA, M. P., ARIFIN, S., HOLIFAH, S., SUGIONO, S., SHOLEH, M. S. & RAMADANI, S. D. 2024. The effect of microplastic contaminated compost on the growth of rice seedlings. *Journal of the Saudi Society of Agricultural Sciences,* 23.
JIANG, Y., ZHOU, S., FEI, J., QIN, Z., YIN, X., SUN, H. & SUN, Y. 2022. Transport of different microplastics in porous media: Effect of the adhesion of surfactants on microplastics. *Water Res,* 215**,** 118262.
JOOS, L. & DE TENDER, C. 2022. Soil under stress: The importance of soil life and how it is influenced by (micro)plastic pollution. *Comput Struct Biotechnol J,* 20**,** 1554-1566.
JU, H., YANG, X., OSMAN, R. & GEISSEN, V. 2023. Effects of microplastics and chlorpyrifos on earthworms (Lumbricus terrestris) and their biogenic transport in sandy soil. *Environmental Pollution,* 316.
KALCIKOVA, G., ZGAJNAR GOTVAJN, A., KLADNIK, A. & JEMEC, A. 2017. Impact of polyethylene microbeads on the floating freshwater plant duckweed Lemna minor. *Environ Pollut,* 230**,** 1108-1115.
KAMP, H., HAEGEN, M. S. V., MANSVELD, W., THUNNISSEN, J., PAUL, R., VRIES, R. J. D., MEIJER, D., OOSTEN, A. M. C. V., POST, H. D., KOLFF, P. H., SUY, G. H. A., MULLEM, D. V., BAAS, B. K. J., HOONDERT, J. S. G., NETELENBOS, T., CUPIDO, D. J. & LEEMANS, E. 2014. Green deal on ship-generated waste.
KAPPLER, A., FISCHER, D., OBERBECKMANN, S., SCHERNEWSKI, G., LABRENZ, M., EICHHORN, K. J. & VOIT, B. 2016. Analysis of environmental microplastics by vibrational microspectroscopy: FTIR, Raman or both? *Anal Bioanal Chem,* 408**,** 8377-8391.
KATSUMI, N., KUSUBE, T., NAGAO, S. & OKOCHI, H. 2021. Accumulation of microcapsules derived from coated fertilizer in paddy fields. *Chemosphere,* 267**,** 129185.
KAWECKI, D. & NOWACK, B. 2019. Polymer-Specific Modeling of the Environmental Emissions of Seven Commodity Plastics As Macro- and Microplastics. *Environmental Science & Technology,* 53.
KELLER, A. S., JIMENEZ-MARTINEZ, J. & MITRANO, D. M. 2020. Transport of Nano- and Microplastic through Unsaturated Porous Media from Sewage Sludge Application. *Environ Sci Technol,* 54**,** 911-920.
KERSHAW, P. & ROCHMAN, C. M. 2016. Source fate and effects of microplastics in the marine environment part 2 of a global assessment.
KHAMBOONRUANG, P., KLORVUTTIMONTARA, S., KANCHANAKETU, T. & PROMMI, T.-O. 2024. Microplastic accumulation in the golden apple snail Pomacea canaliculata serves as a bioindicator in the urban lowlands of Nakhon Pathom Province, central Thailand. *Ecologica Montenegrina,* 80.
KHAN, A. & JIA, Z. 2023. Recent insights into uptake, toxicity, and molecular targets of microplastics and nanoplastics relevant to human health impacts. *iScience,* 26**,** 106061.
KHANDARE, S. D., AGRAWAL, D., MEHRU, N. & CHAUDHARY, D. R. 2022. Marine bacterial based enzymatic degradation of low-density polyethylene (LDPE) plastic. *Journal of Environmental Chemical Engineering,* 10.
KIM, S. K., KIM, J. S., LEE, H. & LEE, H. J. 2021. Abundance and characteristics of microplastics in soils with different agricultural practices: Importance of sources with internal origin and environmental fate. *J Hazard Mater,* 403**,** 123997.
KIM, S. W. & AN, Y.-J. 2019. Soil microplastics inhibit the movement of springtail species. *Environment International,* 126.
KNIAZEV, K., PAVLOVETC, I. M., ZHANG, S., KIM, J., STEVENSON, R. L., DOUDRICK, K. & KUNO, M. 2021. Using Infrared Photothermal Heterodyne Imaging to Characterize Micro- and Nanoplastics in Complex Environmental Matrices. *Environ Sci Technol,* 55**,** 15891-15899.
KOELMANS, A. A., GEBREYOHANES BELAY, B. M., MINTENIG, S. M., MOHAMED NOR, N. H., REDONDO-HASSELERHARM, P. E. & DE RUIJTER, V. N. 2023. Towards a rational and efficient risk assessment for microplastics. *TrAC Trends in Analytical Chemistry,* 165.
KOELMANS, A. A., REDONDO-HASSELERHARM, P. E., MOHAMED NOR, N. H. & KOOI, M. 2020. Solving the Nonalignment of Methods and Approaches Used in Microplastic Research to Consistently Characterize Risk. *Environ Sci Technol,* 54**,** 12307-12315.

KOELMANS, A. A., REDONDO-HASSELERHARM, P. E., NOR, N. H. M., DE RUIJTER, V. N., MINTENIG, S. M. & KOOI, M. 2022. Risk assessment of microplastic particles. *Nature Reviews Materials,* 7**,** 138-152.
KOKALJ, A. J., HORVAT, P., SKALAR, T. & KRŽAN, A. 2018. Plastic bag and facial cleanser derived microplastic do not affect feeding behaviour and energy reserves of terrestrial isopods. *Science of The Total Environment,* 615.
KOUTNIK, V. S., LEONARD, J., EL RASSI, L. A., CHOY, M. M., BRAR, J., GLASMAN, J. B., COWGER, W. & MOHANTY, S. K. 2023. Children's playgrounds contain more microplastics than other areas in urban parks. *Sci Total Environ,* 854**,** 158866.
KUMAR, M. V. & SHEELA, A. M. 2021. Effect of plastic film mulching on the distribution of plastic residues in agricultural fields. *Chemosphere,* 273**,** 128590.
KUNDU, M. N., KOMAKECH, H. C. & LUGOMELA, G. 2022. Analysis of Macro- and Microplastics in Riverine, Riverbanks, and Irrigated Farms in Arusha, Tanzania. *Arch Environ Contam Toxicol,* 82**,** 142-157.
KUNZ, A. 2022. *What is microplastic? Definition of plastic and microplastic* [Online]. Available: https://microplasticresearch.wordpress.com/what-is-microplastic/ [Accessed].
KUTRALAM-MUNIASAMY, G., PÉREZ-GUEVARA, F., ELIZALDE-MARTÍNEZ, I. & SHRUTI, V. C. 2020. Branded milks – Are they immune from microplastics contamination? *Science of The Total Environment,* 714.
LANG, M., WANG, G., YANG, Y., ZHU, W., ZHANG, Y., OUYANG, Z. & GUO, X. 2022. The occurrence and effect of altitude on microplastics distribution in agricultural soils of Qinghai Province, northwest China. *Sci Total Environ,* 810**,** 152174.
LAVELLE, P., DECAËNS, T., AUBERT, M., BAROT, S., BLOUIN, M., BUREAU, F., MARGERIE, P., MORA, P. & ROSSI, J. P. 2006. Soil invertebrates and ecosystem services. *European Journal of Soil Biology,* 42**,** S3-S15.
LEE, J.-Y., CHA, J. & CHIA, R. W. 2022. Current status of researches on microplastics in groundwater and perspectives. *Journal of the Geological Society of Korea,* 58**,** 233-241.
LEE, J.-Y., CHA, J., HA, K. & VIAROLI, S. 2024. Microplastic pollution in groundwater: a systematic review. *Environmental Pollutants and Bioavailability,* 36.
LEHMANN, A., FITSCHEN, K. & RILLIG, M. C. 2019. Abiotic and Biotic Factors Influencing the Effect of Microplastic on Soil Aggregation. *Soil Systems,* 3.
LEITAO, I. A., VAN SCHAIK, L., FERREIRA, A. J. D., ALEXANDRE, N. & GEISSEN, V. 2023. The spatial distribution of microplastics in topsoils of an urban environment - Coimbra city case-study. *Environ Res,* 218**,** 114961.
LI, G., PEI, Z., LI, Y., YANG, R., WANG, P., LIANG, Y., ZHANG, J., ZHANG, Q. & JIANG, G. 2024. A high-precision, effective method for extraction and identification of small-sized microplastics from soil. *Talanta,* 272**,** 125802.
LI, J., PENG, D., OUYANG, Z., LIU, P., FANG, L. & GUO, X. 2022a. Occurrence Status of Microplastics in Main Agricultural Areas of Xinjiang Uygur Autonomous Region, China. *SSRN Electronic Journal*.
LI, J., SONG, Y. & CAI, Y. 2020a. Focus topics on microplastics in soil: Analytical methods, occurrence, transport, and ecological risks. *Environmental Pollution,* 257.
LI, L., LUO, Y., LI, R., ZHOU, Q., PEIJNENBURG, W. J. G. M., YIN, N., YANG, J., TU, C., ZHANG, Y., LI, L., LUO, Y., LI, R., ZHOU, Q., PEIJNENBURG, W. J. G. M., YIN, N., YANG, J., TU, C. & ZHANG, Y. 2020b. Effective uptake of submicrometre plastics by crop plants via a crack-entry mode. *Nature Sustainability 2020,* 3.
LI, L., LUO, Y., PEIJNENBURG, W. J. G. M., LI, R., YANG, J. & ZHOU, Q. 2020c. Confocal measurement of microplastics uptake by plants. *MethodsX,* 7.
LI, L., ZHOU, Q., YIN, N., TU, C., LUO, Y., LI, L., ZHOU, Q., YIN, N., TU, C. & LUO, Y. 2019a. Uptake and accumulation of microplastics in an edible plant. *Chinese Science Bulletin,* 64.
LI, L. Z., LUO, Y. M., LI, R. J., ZHOU, Q., PEIJNENBURG, W. J. G. M., YIN, N., YANG, J., TU, C. & ZHANG, Y. C. 2020d. Effective uptake of submicrometre plastics by crop plants via a crack-entry mode. *Nature Sustainability,* 3.
LI, Q., WU, J., ZHAO, X., GU, X. & JI, R. 2019b. Separation and identification of microplastics from soil and sewage sludge. *Environ Pollut,* 254**,** 113076.
LI, S., DING, F., FLURY, M., WANG, Z., XU, L., LI, S., JONES, D. L. & WANG, J. 2022b. Macro- and microplastic accumulation in soil after 32 years of plastic film mulching. *Environ Pollut,* 300**,** 118945.

LI, S., WANG, T., GUO, J., DONG, Y., WANG, Z., GONG, L. & LI, X. 2021. Polystyrene microplastics disturb the redox homeostasis, carbohydrate metabolism and phytohormone regulatory network in barley. *J Hazard Mater,* 415**,** 125614.

LI, W., WUFUER, R., DUO, J., WANG, S., LUO, Y., ZHANG, D. & PAN, X. 2020e. Microplastics in agricultural soils: Extraction and characterization after different periods of polythene film mulching in an arid region. *Science of The Total Environment,* 749.

LI, X., WANG, R., DAI, W., LUAN, Y. & LI, J. 2023a. Impacts of Micro(nano)plastics on Terrestrial Plants: Germination, Growth, and Litter. *Plants (Basel),* 12.

LI, X., WANG, R., DAI, W., LUAN, Y., LI, J., LI, X., WANG, R., DAI, W., LUAN, Y. & LI, J. 2023b. Impacts of Micro(nano)plastics on Terrestrial Plants: Germination, Growth, and Litter. *Plants 2023, Vol. 12, Page 3554,* 12.

LI, X., WANG, X., REN, C., PALANSOORIYA, K. N., WANG, Z. & CHANG, S. X. 2023c. Microplastic pollution: Phytotoxicity, environmental risks, and phytoremediation strategies. *Critical Reviews in Environmental Science and Technology,* 54**,** 486-507.

LI, X., ZHANG, T., LV, W., WANG, H., CHEN, H., XU, Q., CAI, H. & DAI, J. 2022c. Intratracheal administration of polystyrene microplastics induces pulmonary fibrosis by activating oxidative stress and Wnt/β-catenin signaling pathway in mice. *Ecotoxicology and Environmental Safety,* 232.

LI, Y., PENG, L., FU, J., DAI, X. & WANG, G. 2022d. A microscopic survey on microplastics in beverages: the case of beer, mineral water and tea. *Analyst,* 147.

LI, Z., WANG, X., LIANG, S., LI, H. & SUN, L. 2023d. Pyr-GC-MS analysis of microplastics extracted from farmland soils. *International Journal of Environmental Analytical Chemistry,* 103.

LI, Z., XIE, Y., ZENG, Y., ZHANG, Z., SONG, Y., HONG, Z., MA, L., HE, M., MA, H. & CUI, F. 2022e. Plastic leachates lead to long-term toxicity in fungi and promote biodegradation of heterocyclic dye. *Science of The Total Environment,* 806.

LIANG, Y., LEHMANN, A., YANG, G., LEIFHEIT, E. F. & RILLIG, M. C. 2021. Effects of Microplastic Fibers on Soil Aggregation and Enzyme Activities Are Organic Matter Dependent. *Frontiers in Environmental Science,* 9.

LIU, H., YANG, X., LIU, G., LIANG, C., XUE, S., CHEN, H., RITSEMA, C. J. & GEISSEN, V. 2017. Response of soil dissolved organic matter to microplastic addition in Chinese loess soil. *Chemosphere,* 185.

LIU, M., LU, S., SONG, Y., LEI, L., HU, J., LV, W., ZHOU, W., CAO, C., SHI, H., YANG, X. & HE, D. 2018. Microplastic and mesoplastic pollution in farmland soils in suburbs of Shanghai, China. *Environ Pollut,* 242**,** 855-862.

LIU, M., SONG, Y., LU, S., QIU, R., HU, J., LI, X., BIGALKE, M., SHI, H. & HE, D. 2019. A method for extracting soil microplastics through circulation of sodium bromide solutions. *Sci Total Environ,* 691**,** 341-347.

LIU, X., LIN, H., XU, S., YAN, Y., YU, R. & HU, G. 2023a. Occurrence, distribution, and characteristics of microplastics in agricultural soil around a solid waste treatment center in southeast China. *J Soils Sediments,* 23**,** 936-946.

LIU, X., TANG, N., YANG, W. & CHANG, J. 2022a. Microplastics pollution in the soils of various land-use types along Sheshui River basin of Central China. *Sci Total Environ,* 806**,** 150620.

LIU, Y., GUO, R., ZHANG, S., SUN, Y. & WANG, F. 2022b. Uptake and translocation of nano/microplastics by rice seedlings: Evidence from a hydroponic experiment. *Journal of Hazardous Materials,* 421.

LIU, Y., HU, W., HUANG, Q., QIN, J., ZHENG, Y., WANG, J., LI, X., WANG, Q., GUO, G. & HU, S. 2022c. Plastic mulch debris in rhizosphere: Interactions with soil-microbe-plant systems. *Science of The Total Environment,* 807.

LIU, Y., HU, J.W., HUANG,Y., KREKELBERGH, N., KUSUMAWARDANI, P.N., SLEUTEL, S., PARAKHONSKIY, B.V., HOOGENBOOM, R., SKIRTACH, A., DE NEVE, S. Under review. Magnetic labelling and extraction of micrometer-sized microplastics from soil. *J Hazard Mater*.

LIU, Y., LI, J., PARAKHONSKIY, B. V., HOOGENBOOM, R., SKIRTACH, A. & DE NEVE, S. 2024. Labelling of micro- and nanoplastics for environmental studies: state-of-the-art and future challenges. *J Hazard Mater,* 462**,** 132785.

LIU, Y., ZHONG, Y., HU, C., XIAO, M., DING, F., YU, Y., YAO, H., ZHU, Z., CHEN, J., GE, T. & DING, J. 2023b. Distribution of microplastics in soil aggregates after film mulching. *Soil Ecology Letters,* 5.

LJUNG, E., OLESEN, K. B., ANDERSSON, P.-G., FÄLTSTRÖM, E., VOLLERTSEN, J., WITTGREN, H. B. & HAGMAN, M. 2018. Mikroplaster i kretsloppet. *In:* UTVECKLING, S. V. (ed.).

LÖDER, M. G. J., KUCZERA, M., MINTENIG, S., LORENZ, C. & GERDTS, G. 2015. Focal plane array detector-based micro-Fourier-transform infrared imaging for the analysis of microplastics in environmental samples. *Environmental Chemistry,* 12.

LU, Y., LI, M. C., LEE, J., LIU, C. & MEI, C. 2023. Microplastic remediation technologies in water and wastewater treatment processes: Current status and future perspectives. *Sci Total Environ,* 868**,** 161618.

LUO, X., WANG, Z., YANG, L., GAO, T. & ZHANG, Y. 2022. A review of analytical methods and models used in atmospheric microplastic research. *Science of The Total Environment,* 828.

LWANGA, E. H., VAN ROSHUM, I., MUNHOZ, D. R., MENG, K., REZAEI, M., GOOSSENS, D., BIJSTERBOSCH, J., ALEXANDRE, N., OOSTERWIJK, J., KROL, M., PETERS, P., GEISSEN, V. & RITSEMA, C. 2023. Microplastic appraisal of soil, water, ditch sediment and airborne dust: The case of agricultural systems. *Environ Pollut,* 316**,** 120513.

MADDELA, N. R., KAKARLA, D., VENKATESWARLU, K. & MEGHARAJ, M. 2023. Additives of plastics: Entry into the environment and potential risks to human and ecological health. *J Environ Manage,* 348**,** 119364.

MARTINDALE, S., WEISBERG, S. B. & COFFIN, S. 2020. Status of Legislation and Regulatory Drivers for Microplastics in California

Institution: Southern California Coastal Water Research Project Authority.

MASON, S. A., WELCH, V. G. & NERATKO, J. 2018. Synthetic Polymer Contamination in Bottled Water. *Front Chem,* 6**,** 407.

MASURA, J., BAKER, J., FOSTER, G., ARTHUR, C. & HERRING, C. 2015. Laboratory Methods for the Analysis of Microplastics inthe Marine Environment: Recommendations for quantifying synthetic particles in waters and sediments. *NOAA Marine Debris Program*.

MATEO-SAGASTA, J., MEDLICOTT, K., QADIR, M., RASCHID-SALLY, L. & DRECHSEL, P. Proceedings of the UN-water project on the Safe use of wastewater in agriculture. 2013. UNU-INWEH.

MCDEVITT, J. P., CRIDDLE, C. S., MORSE, M., HALE, R. C., BOTT, C. B. & ROCHMAN, C. M. 2017. Addressing the Issue of Microplastics in the Wake of the Microbead-Free Waters Act-A New Standard Can Facilitate Improved Policy. *Environ Sci Technol,* 51**,** 6611-6617.

MEIXNER, K., KUBICZEK, M. & FRITZ, I. 2020. Microplastic in soil–current status in Europe with special focus on method tests with Austrian samples. *AIMS Environmental Science,* 7**,** 174-191.

MENGISTU, D., NILSEN, V., COUTRIS, C., AMDAL, H. M. & HEISTAD, A. 2022. Microplastics Concentrations in Soil Along a Racetrack. *Water, Air, & Soil Pollution,* 234.

MIAO, L., WANG, P., HOU, J., YAO, Y., LIU, Z., LIU, S. & LI, T. 2019. Distinct community structure and microbial functions of biofilms colonizing microplastics. *Sci Total Environ,* 650**,** 2395-2402.

MICROBEAD-FREE WATERS ACT 2015. H.R.1321 - 114th Congress (2015-2016): Microbead-Free Waters Act of 2015. *H.R.1321.*

MINTENIG, S. M., LODER, M. G. J., PRIMPKE, S. & GERDTS, G. 2019. Low numbers of microplastics detected in drinking water from ground water sources. *Sci Total Environ,* 648**,** 631-635.

MOKHTARZADEH, Z., KESHAVARZI, B., MOORE, F., BUSQUETS, R., REZAEI, M., PADOAN, E. & AJMONE-MARSAN, F. 2022. Microplastics in industrial and urban areas in South-West Iran. *International Journal of Environmental Science and Technology,* 19**,** 10199-10210.

MÖLLER, J. N., LODER, M. G. J. & LAFORSCH, C. 2020. Finding Microplastics in Soils: A Review of Analytical Methods. *Environ Sci Technol,* 54**,** 2078-2090.

MONIRA, S., ROYCHAND, R., BHUIYAN, M. A., HAI, F. I. & PRAMANIK, B. K. 2022. Identification, classification and quantification of microplastics in road dust and stormwater. *Chemosphere,* 299**,** 134389.

MORENO-JIMÉNEZ, E., LEIFHEIT, E. F., PLAZA, C., FENG, L., BERGMANN, J., WULF, A., LEHMANN, A. & RILLIG, M. C. 2022. Effects of microplastics on crop nutrition in fertile soils and interaction with arbuscular mycorrhizal fungi. *Journal of Sustainable Agriculture and Environment,* 1.

MP-SED 2023. Dredging operations and environmental research. *In:* ENVIRONMENTAL LABORATORY, U. A. E. R. A. D. (ed.).

MU, H., WANG, Y., ZHANG, H., GUO, F., LI, A., ZHANG, S., LIU, S. & LIU, T. 2022. High abundance of microplastics in groundwater in Jiaodong Peninsula, China. *Sci Total Environ,* 839**,** 156318.
MUELLER, C. W., SCHLUND, S., PRIETZEL, J., KÖGEL-KNABNER, I. & GUTSCH, M. 2012. Soil aggregate destruction by ultrasonication increases soil organic matter mineralization and mobility. *Soil Biology and Biochemistry,* 76:1634-1643.
MÜHLSCHLEGEL, P., HAUK, A., WALTER, U. & SIEBER, R. 2017. Lack of evidence for microplastic contamination in honey. *Food additives & contaminants. Part A, Chemistry, analysis, control, exposure & risk assessment,* 34.
MULLER, A., KOCHER, B., ALTMANN, K. & BRAUN, U. 2022. Determination of tire wear markers in soil samples and their distribution in a roadside soil. *Chemosphere,* 294**,** 133653.
MUNHOZ, D. R., HARKES, P., BERIOT, N., LARRETA, J. & BASURKO, O. C. 2022. Microplastics: A Review of Policies and Responses. *Microplastics,* 2**,** 1-26.
NADERI BENI, N., KARIMIFARD, S., GILLEY, J., MESSER, T., SCHMIDT, A. & BARTELT-HUNT, S. 2023. Higher concentrations of microplastics in runoff from biosolid-amended croplands than manure-amended croplands. *Communications Earth & Environment,* 4.
NEMATOLLAHI, M. J., KESHAVARZI, B., MOHIT, F., MOORE, F. & BUSQUETS, R. 2022. Microplastic occurrence in urban and industrial soils of Ahvaz metropolis: A city with a sustained record of air pollution. *Sci Total Environ,* 819**,** 152051.
NGUYEN, M. K., LIN, C., HUNG, N. T. Q., VO, D. N., NGUYEN, K. N., THUY, B. T. P., HOANG, H. G. & TRAN, H. T. 2022. Occurrence and distribution of microplastics in peatland areas: A case study in Long An province of the Mekong Delta, Vietnam. *Sci Total Environ,* 844**,** 157066.
NI, B.-J., ZHU, Z.-R., LI, W.-H., YAN, X., WEI, W., XU, Q., XIA, Z., DAI, X. & SUN, J. 2020. Microplastics Mitigation in Sewage Sludge through Pyrolysis: The Role of Pyrolysis Temperature. *Environmental Science & Technology Letters,* 7**,** 961-967.
NIU, H., LIU, S., JIANG, Y., HU, Y., LI, Y., HE, L., XING, M., LI, X., WU, L., CHEN, Z., WANG, X. & LOU, X. 2023. Are Microplastics Toxic? A Review from Eco-Toxicity to Effects on the Gut Microbiota. *Metabolites,* 13.
NIZAMALI, J., MINTENIG, S. M. & KOELMANS, A. A. 2023. Assessing microplastic characteristics in bottled drinking water and air deposition samples using laser direct infrared imaging. *Journal of Hazardous Materials,* 441.
NUELLE, M. T., DEKIFF, J. H., REMY, D. & FRIES, E. 2014. A new analytical approach for monitoring microplastics in marine sediments. *Environ Pollut,* 184**,** 161-9.
O'CONNOR, D., PAN, S., SHEN, Z., SONG, Y., JIN, Y., WU, W. M. & HOU, D. 2019. Microplastics undergo accelerated vertical migration in sand soil due to small size and wet-dry cycles. *Environ Pollut,* 249**,** 527-534.
OKEKE, E. S., CHUKWUDOZIE, K. I., ADDEY, C. I., OKORO, J. O., CHIDIKE EZEORBA, T. P., ATAKPA, E. O., OKOYE, C. O. & NWUCHE, C. O. 2023. Micro and nanoplastics ravaging our agroecosystem: A review of occurrence, fate, ecological impacts, detection, remediation, and prospects. *Heliyon,* 9**,** e13296.
OMIDOYIN, K. C. & JHO, E. H. 2023. Effect of microplastics on soil microbial community and microbial degradation of microplastics in soil: A review. *Environmental Engineering Research,* 28**,** 220716-0.
PAÇO, A., JACINTO, J., DA COSTA, J. P., SANTOS, P. S. M., VITORINO, R., DUARTE, A. C. & ROCHA-SANTOS, T. 2018. Biotechnological tools for the effective management of plastics in the environment. *Critical Reviews in Environmental Science and Technology,* 49**,** 410-441.
PALAZOT, M., SOCCALINGAME, L., FROGER, C., JOLIVET, C., BISPO, A., KEDZIERSKI, M. & BRUZAUD, S. 2024. First national reference of microplastic contamination of French soils. *Sci Total Environ,* 918**,** 170564.
PARK, S. Y. & KIM, C. G. 2022. A comparative study on the distribution behavior of microplastics through FT-IR analysis on different land uses in agricultural soils. *Environ Res,* 215**,** 114404.
PAYNE, D. 1988. *Soil structure, tilth and mechanical behaviour.*
PEÑA, A., RODRÍGUEZ-LIÉBANA, J. A. & DELGADO-MORENO, L. 2023. Interactions of Microplastics with Pesticides in Soils and Their Ecotoxicological Implications. *Agronomy,* 13.
PENG, C., WANG, J., LIU, X. & WANG, L. 2022. Differences in the Plastispheres of Biodegradable and Non-biodegradable Plastics: A Mini Review. *Front Microbiol,* 13**,** 849147.

PERFETTI-BOLAÑO, A., ARANEDA, A., MUÑOZ, K. & BARRA, R. O. 2022. Occurrence and Distribution of Microplastics in Soils and Intertidal Sediments at Fildes Bay, Maritime Antarctica. *Frontiers in Marine Science,* 8.

PETTIPAS, S., BERNIER, M. & WALKER, T. R. 2016. A Canadian policy framework to mitigate plastic marine pollution. *Marine Policy,* 68**,** 117-122.

PIEHL, S., LEIBNER, A., LODER, M. G. J., DRIS, R., BOGNER, C. & LAFORSCH, C. 2018. Identification and quantification of macro- and microplastics on an agricultural farmland. *Sci Rep,* 8**,** 17950.

PIGNATTELLI, S., BROCCOLI, A. & RENZI, M. 2020. Physiological responses of garden cress (L. sativum) to different types of microplastics. *Sci Total Environ,* 727**,** 138609.

PITT, J. A., KOZAL, J. S., JAYASUNDARA, N., MASSARSKY, A., TREVISAN, R., GEITNER, N., WIESNER, M., LEVIN, E. D. & GIULIO, R. T. D. 2018. Uptake, tissue distribution, and toxicity of polystyrene nanoparticles in developing zebrafish (Danio rerio). *Aquatic Toxicology,* 194.

PIVNENKO, K., GRANBY, K., ERIKSSON, E. & ASTRUP, T. F. 2017. Recycling of plastic waste: Screening for brominated flame retardants (BFRs). *Waste Management,* 69.

PLASTIC REDUCTION AND CIRCULAR ECONOMY ACT 2021. Plastic Reduction and Circular Economy Act 2021. New South Wales.

PLASTICS EUROPE 2024. Plastics - the fast Facts.

PLASTICSEUROPE 2019. Plastics - the Facts 2019: An analysis of European plastics production, demand and waste data. PlasticsEurope.

PLASTICSEUROPE 2024. The circular economy for plastics - A European analysis.

PRATA, J. C., DA COSTA, J. P., DUARTE, A. C. & ROCHA-SANTOS, T. 2019. Methods for sampling and detection of microplastics in water and sediment: A critical review. *TrAC Trends in Analytical Chemistry,* 110**,** 150-159.

PUYPE, F., SAMSONEK, J., KNOOP, J., EGELKRAUT-HOLTUS, M. & ORTLIEB, M. 2015. Evidence of waste electrical and electronic equipment (WEEE) relevant substances in polymeric food-contact articles sold on the European market. *Food Additives & Contaminants: Part A*.

QIU, Y., ZHOU, S., ZHANG, C., QIN, W., LV, C. & ZOU, M. 2023. Identification of potentially contaminated areas of soil microplastic based on machine learning: A case study in Taihu Lake region, China. *Sci Total Environ,* 877**,** 162891.

QUIK, J. T. K., FABER, M., RIDDER, J. & WAAIJERS-VAN DER LOOP, S. L. 2022. Wat weten we over microplastics in het milieu? Kennisagenda microplastics in het milieu. RIVM.

QUIK, J. T. K., HIDS, A. R., STEENMEIJER, M. A., MELLINK, Y. & BRUGGEN, A. V. 2024. Emission of Microplastics to Water, Soil, and Air What can we do about it?

RADFORD, F., HORTON, A., HUDSON, M., SHAW, P. & WILLIAMS, I. 2023. Agricultural soils and microplastics: Are biosolids the problem? *Frontiers in Soil Science,* 2.

RAFIQUE, A., IRFAN, M., MUMTAZ, M. & QADIR, A. 2020. Spatial distribution of microplastics in soil with context to human activities: a case study from the urban center. *Environ Monit Assess,* 192**,** 671.

RAGOOBUR, D., HUERTA-LWANGA, E. & SOMAROO, G. D. 2021. Microplastics in agricultural soils, wastewater effluents and sewage sludge in Mauritius. *Sci Total Environ,* 798**,** 149326.

RAGUSA, A., SVELATO, A., SANTACROCE, C., CATALANO, P., NOTARSTEFANO, V., CARNEVALI, O., PAPA, F., RONGIOLETTI, M. C. A., BAIOCCO, F., DRAGHI, S., D'AMORE, E., RINALDO, D., MATTA, M. & GIORGINI, E. 2021. Plasticenta: First evidence of microplastics in human placenta. *Environment International,* 146.

RAMAGE, S., PAGALING, E., HAGHI, R. K., DAWSON, L. A., YATES, K., PRABHU, R., HILLIER, S. & DEVALLA, S. 2022. Rapid extraction of high- and low-density microplastics from soil using high-gradient magnetic separation. *Sci Total Environ,* 831**,** 154912.

RAMBABU, K., BHARATH, G., GOVARTHANAN, M., KUMAR, P. S., SHOW, P. L. & BANAT, F. 2023. Bioprocessing of plastics for sustainable environment: Progress, challenges, and prospects. *TrAC Trends in Analytical Chemistry,* 166.

RAZA, M., LEE, J. Y. & CHA, J. 2022. Microplastics in soil and freshwater: Understanding sources, distribution, potential impacts, and regulations for management. *Sci Prog,* 105**,** 368504221126676.

RAZAVIARANI, V., SAUDAGAR, A., GALLAGE, S., SHRINATH, S. & ARAB, G. 2024. Comprehensive investigation on microplastics from source to sink. *Clean Technologies and Environmental Policy,* 26**,** 1755-1782.

RE, V. 2019. Shedding light on the invisible: addressing the potential for groundwater contamination by plastic microfibers. *Hydrogeology Journal,* 27**,** 2719-2727.

REDONDO-HASSELERHARM, P., RICO, A., HUERTA LWANGA, E., VAN GESTEL, C. & KOELMANS, A. 2024. Source-specific probabilistic risk assessment of microplastics in soils applying quality criteria and data alignment methods - PubMed. *Journal of hazardous materials,* 467.

REDONDO-HASSELERHARM, P. E., RICO, A. & KOELMANS, A. A. 2023. Risk assessment of microplastics in freshwater sediments guided by strict quality criteria and data alignment methods. *Journal of Hazardous Materials,* 441.

REN, Z., GUI, X., XU, X., ZHAO, L., QIU, H. & CAO, X. 2021. Microplastics in the soil-groundwater environment: Aging, migration, and co-transport of contaminants - A critical review. *J Hazard Mater,* 419**,** 126455.

RENZI, M., SPECCHIULLI, A., BLAŠKOVIĆ, A., MANZO, C., MANCINELLI, G., CILENTI, L., RENZI, M., SPECCHIULLI, A., BLAŠKOVIĆ, A., MANZO, C., MANCINELLI, G. & CILENTI, L. 2018. Marine litter in stomach content of small pelagic fishes from the Adriatic Sea: sardines (Sardina pilchardus) and anchovies (Engraulis encrasicolus). *Environmental Science and Pollution Research 2018 26:3,* 26.

REZAEI, M., ABBASI, S., POURMAHMOOD, H., OLESZCZUK, P., RITSEMA, C. & TURNER, A. 2022. Microplastics in agricultural soils from a semi-arid region and their transport by wind erosion. *Environ Res,* 212**,** 113213.

RILLIG, M. C. 2012. Microplastic in terrestrial ecosystems and the soil? *Environ Sci Technol,* 46**,** 6453-4.

RILLIG, M. C., INGRAFFIA, R. & DE SOUZA MACHADO, A. A. 2017a. Microplastic Incorporation into Soil in Agroecosystems. *Front Plant Sci,* 8**,** 1805.

RILLIG, M. C., LEIFHEIT, E. & LEHMANN, J. 2021. Microplastic effects on carbon cycling processes in soils. *PLoS Biol,* 19**,** e3001130.

RILLIG, M. C., ZIERSCH, L. & HEMPEL, S. 2017b. Microplastic transport in soil by earthworms. *Sci Rep,* 7**,** 1362.

RIVM. 2024. *Main sources of microplastics in the environment are tyre erosion, plastic waste and industrial plastic granules* [Online]. Available: https://www.rivm.nl/en/news/main-sources-of-microplastics-in-environment-are-tyre-erosion-plastic-waste-and-industrial#:~:text=The%20main%20sources%20of%20these,the%20air%20and%20in%20water. [Accessed 28/02/2025].

RIZOS, V., URBAN, P., RIGHETTI, E. & KASSAB, A. 2023. Chemical recycling of plastics. CEPS.

ROBERTS, J. A., MICHELE, G. D., PERNIGOTTO, G., GASPARELLA, A. & AVESANI, S. 2022. Impact of active façade control parameters and sensor network complexity on comfort and efficiency: A residential Italian case-study. *Energy and Buildings,* 255.

RODRIGUES, M., ABRANTES, N., GONÇALVES, F., NOGUEIRA, H., MARQUES, J. & GONÇALVES, A. 2018. Spatial and temporal distribution of microplastics in water and sediments of a freshwater system (Antuã River, Portugal). *The Science of the total environment,* 633.

ROLSKY, C., KELKAR, V., DRIVER, E. & HALDEN, R. U. 2020. Municipal sewage sludge as a source of microplastics in the environment. *Current Opinion in Environmental Science & Health,* 14**,** 16-22.

ROSATO, A., BARONE, M., NEGRONI, A., BRIGIDI, P., FAVA, F., BIAGI, E., CANDELA, M. & ZANAROLI, G. 2022. Bacterial colonization dynamics of different microplastic types in an anoxic salt marsh sediment and impact of adsorbed polychlorinated biphenyls on the plastisphere. *Environ Pollut,* 315**,** 120411.

RÜCKNAGEL, J., CHRISTEN, O., HOFMANN, B. & ULRICH, S. 2012. A simple model to estimate change in precompression stress as a function of water content on the basis of precompression stress at field capacity. *Geoderma,* 177-178**,** 1-7.

SA'ADU, I. & FARSANG, A. 2022. Greenhouse farming as a source of macroplastic and microplastics contamination in agricultural soils: a case study from Southeast-Hungary. *Agrokémia és Talajtan,* 71**,** 43-57.

SAEEDI, M. 2023. How microplastics interact with food chain: a short overview of fate and impacts. *J Food Sci Technol,* 61**,** 1-11.

SAINI, A. & SHARMA, J. G. 2022. Emerging microplastic contamination in ecosystem: An urge for environmental sustainability. *Journal of Applied Biology & Biotechnology***,** 66-75.

SALEEM, J., TAHIR, F., BAIG, M. Z. K., AL-ANSARI, T. & MCKAY, G. 2023. Assessing the environmental footprint of recycled plastic pellets: A life-cycle assessment perspective. *Environmental Technology & Innovation,* 32.

SAMAK, N. A., JIA, Y., SHARSHAR, M. M., MU, T., YANG, M., PEH, S. & XING, J. 2020. Recent advances in biocatalysts engineering for polyethylene terephthalate plastic waste green recycling. *Environment International,* 145.

SAMANDRA, S., JOHNSTON, J. M., JAEGER, J. E., SYMONS, B., XIE, S., CURRELL, M., ELLIS, A. V. & CLARKE, B. O. 2022. Microplastic contamination of an unconfined groundwater aquifer in Victoria, Australia. *Sci Total Environ,* 802**,** 149727.

SAMSONEK, J. & PUYPE, F. 2013. Occurrence of brominated flame retardants in black thermo cups and selected kitchen utensils purchased on the European market. *Food Additives & Contaminants: Part A*.

SANTACRUZ-JUÁREZ, E., BUENDIA-CORONA, R. E., RAMÍREZ, R. E. & SÁNCHEZ, C. 2021. Fungal enzymes for the degradation of polyethylene: Molecular docking simulation and biodegradation pathway proposal. *Journal of Hazardous Materials,* 411.

SANTOS, L., INSA, S., ARXE, M., BUTTIGLIERI, G., RODRIGUEZ-MOZAZ, S. & BARCELO, D. 2023. Analysis of microplastics in the environment: Identification and quantification of trace levels of common types of plastic polymers using pyrolysis-GC/MS. *MethodsX,* 10**,** 102143.

SARWAR, N., IMRAN, M., SHAHEEN, M. R., ISHAQUE, W., KAMRAN, M. A., MATLOOB, A., REHIM, A. & HUSSAIN, S. 2017. Phytoremediation strategies for soils contaminated with heavy metals: Modifications and future perspectives. *Chemosphere,* 171**,** 710-721.

SCHADE, A., MELZER, M., ZIMMERMANN, S., SCHWARZ, T., STOEWE, K. & KUHN, H. 2024. Plastic Waste Recycling–A Chemical Recycling Perspective. *ACS Sustainable Chemistry & Engineering,* 12.

SCHELL, T., HURLEY, R., BUENAVENTURA, N. T., MAURI, P. V., NIZZETTO, L., RICO, A. & VIGHI, M. 2022. Fate of microplastics in agricultural soils amended with sewage sludge: Is surface water runoff a relevant environmental pathway? *Environ Pollut,* 293**,** 118520.

SCHEURER, M. & BIGALKE, M. 2018. Microplastics in Swiss Floodplain Soils. *Environ Sci Technol,* 52**,** 3591-3598.

SCHÖPFER, L., MENZEL, R., SCHNEPF, U., RUESS, L., MARHAN, S., BRÜMMER, F., PAGEL, H. & KANDELER, E. 2020. Microplastics Effects on Reproduction and Body Length of the Soil-Dwelling Nematode Caenorhabditis elegans. *Frontiers in Environmental Science,* 8.

SCHÖPFER, L., MÖLLER, J. N., STEINER, T., SCHNEPF, U., MARHAN, S., RESCH, J., BAYHA, A., LÖDER, M. G. J., FREITAG, R., BRÜMMER, F., LAFORSCH, C., STRECK, T., FORBERGER, J., KRANERT, M., KANDELER, E. & PAGEL, H. 2022. Microplastics persist in an arable soil but do not affect soil microbial biomass, enzyme activities, and crop yield. *Journal of Plant Nutrition and Soil Science,* 185**,** 836-849.

SCHWABL, P., KÖPPEL, S., KÖNIGSHOFER, P., BUCSICS, T., TRAUNER, M., REIBERGER, T. & LIEBMANN, B. 2019. Detection of Various Microplastics in Human Stool: A Prospective Case Series. *Annals of internal medicine,* 171.

SCOPETANI, C., CHELAZZI, D., MIKOLA, J., LEINIO, V., HEIKKINEN, R., CINCINELLI, A. & PELLINEN, J. 2020. Olive oil-based method for the extraction, quantification and identification of microplastics in soil and compost samples. *Sci Total Environ,* 733**,** 139338.

SEMMOURI, I., VERCAUTEREN, M., VAN ACKER, E., PEQUEUR, E., ASSELMAN, J. & JANSSEN, C. 2023. Distribution of microplastics in freshwater systems in an urbanized region: A case study in Flanders (Belgium). *Science of The Total Environment,* 872.

SHAFEA, L., YAP, J., BERIOT, N., FELDE, V. J. M. N. L., OKOFFO, E. D., ENYOH, C. E. & PETH, S. 2023. Microplastics in agroecosystems: A review of effects on soil biota and key soil functions. *Journal of Plant Nutrition and Soil Science,* 186.

SHARMA, S., BASU, S., SHETTI, N., NADAGOUDA, M. & AMINABHAVI, T. 2021. Microplastics in the environment: Occurrence, perils, and eradication - PubMed. *Chemical engineering journal (Lausanne, Switzerland : 1996),* 408.

SHARUDDIN, S. D. A., ABNISA, F., WAN, M. A. W. D. & MOHAMED, K. T. 2016. A review on pyrolysis of plastic wastes. *Energy Conversion and Management,* 115**,** 308-326.

SHI, J., WANG, J., LV, J., WANG, Z., PENG, Y. & WANG, X. 2022a. Microplastic presence significantly alters soil nitrogen transformation and decreases nitrogen bioavailability under contrasting temperatures. *J Environ Manage,* 317**,** 115473.

SHI, R., LIU, W., LIAN, Y., WANG, Q., ZEB, A. & TANG, J. 2022b. Phytotoxicity of polystyrene, polyethylene and polypropylene microplastics on tomato (Lycopersicon esculentum L.). *J Environ Manage,* 317**,** 115441.

SHI, X., ZHANG, X., GAO, W., ZHANG, Y. & HE, D. 2022c. Removal of microplastics from water by magnetic nano-Fe(3)O(4). *Sci Total Environ,* 802**,** 149838.

SHIM, W. J., HONG, S. H. & EO, S. E. 2017. Identification methods in microplastic analysis: a review. *Analytical Methods,* 9**,** 1384-1391.

SIEBER, R., KAWECKI, D. & NOWACK, B. 2020. Dynamic probabilistic material flow analysis of rubber release from tires into the environment - PubMed. *Environmental pollution,* 258.

SINGH, B. S. M., SINGH, D. & DHAL, N. K. 2022. Enhanced phytoremediation strategy for sustainable management of heavy metals and radionuclides. *Case Studies in Chemical and Environmental Engineering,* 5.

SINGH, N., WALKER, T. R., SINGH, N. & WALKER, T. R. 2024. Plastic recycling: A panacea or environmental pollution problem. *npj Materials Sustainability 2024 2:1,* 2.

SIX, J., BOSSUYT, H., DEGRYZE, S. & DENEF, K. 2004. A history of research on the link between (micro)aggregates, soil biota, and soil organic matter dynamics. *Soil and Tillage Research,* 79**,** 7-31.

STAPLETON, M. J., ANSARI, A. J., AHMED, A. & HAI, F. I. 2023. Evaluating the generation of microplastics from an unlikely source: The unintentional consequence of the current plastic recycling process. *Science of The Total Environment,* 902.

STATBEL. 2022. *Afvalproductie* [Online]. Available: https://statbel.fgov.be/nl/themas/leefmilieu/afval-en-vervuiling/afvalproductie [Accessed 28/02/2025].

STEGMANN, P., LENSEN, S., HERLAAR, S., SCHWARZ, A. & VVAN DEN BEUKEN, E. 2024. Circularity and greenhouse gas assessment of the plastic packaging and beverage carton system in the Netherlands until 2050 For a business-as-usual and a circular scenario. *In:* TNO (ed.).

STEINMETZ, Z., WOLLMANN, C., SCHAEFER, M., BUCHMANN, C., DAVID, J., TROGER, J., MUNOZ, K., FROR, O. & SCHAUMANN, G. E. 2016. Plastic mulching in agriculture. Trading short-term agronomic benefits for long-term soil degradation? *Sci Total Environ,* 550**,** 690-705.

SU, L., NAN, B., CRAIG, N. J. & PETTIGROVE, V. 2020. Temporal and spatial variations of microplastics in roadside dust from rural and urban Victoria, Australia: Implications for diffuse pollution. *Chemosphere,* 252**,** 126567.

SU, X., LIU, M., YUAN, J., HUANG, X., LU, Z., XU, J. & HE, Y. 2023. Potential risk of co-occurrence of microplastics and chlorinated persistent organic pollutants to coastal wetlands: Evidence from a case study. *Environ Pollut,* 320**,** 121087.

SUNDT, P., SCHULZE, S. & SYVERSEN, F. 2014. Sources of microplastic pollution to the marine environment. *In:* AGENCY, M. F. T. N. E. (ed.).

TAJWAR, M., SHREYA, S. S., HASAN, M., HOSSAIN, M. B., GAZI, M. Y. & SAKIB, N. 2022. Assessment of microplastics as contaminants in a coal mining region. *Heliyon,* 8**,** e11666.

TALVITIE, J. & HEINONEN, M. 2014. Preliminary Study on Synthetic Microfibers and Particles at a Municipal Waste Water Treatment Plant. *Baltic Marine Environmental Protection Commission HELCOM***,** 1-14.

TANG, S., GAO, L., ZHAO, T. & TIAN, A. 2024. Enhancing the removal efficiency of microplastics in drinking water treatment. *Journal of Water Process Engineering,* 57.

TAYLOR, S. E., PEARCE, C. I., SANGUINET, K. A., HU, D., CHRISLER, W. B., KIM, Y.-M., WANG, Z. & FLURY, M. 2020. Polystyrene nano- and microplastic accumulation at Arabidopsis and wheat root cap cells, but no evidence for uptake into roots. *Environmental Science: Nano,* 7**,** 1942-1953.

THAKUR, B., SINGH, J., SINGH, J., ANGMO, D. & VIG, A. P. 2023. Biodegradation of different types of microplastics: Molecular mechanism and degradation efficiency. *Sci Total Environ,* 877**,** 162912.

THAPLIYAL, C., PRIYA, A., SINGH, S. B., BAHUGUNA, V. & DAVEREY, A. 2024. Potential strategies for bioremediation of microplastic contaminated soil. *Environmental Chemistry and Ecotoxicology,* 6**,** 117-131.

THOMAS, B., VINKA, C., PAWAN, L. & DAVID, S. 2022. Sustainable groundwater treatment technologies for underserved rural communities in emerging economies. *Sci Total Environ,* 813**,** 152633.
THOMAS, N. L., CLARKE, J., MCLAUCHLIN, A. R. & PATRICK, S. G. 2012. Oxodegradable plastics: degradation, environmental impact and recycling. *Proceedings of the Institution of Civil Engineers - Waste and Resource Management,* 165**,** 133-140.
THOMPSON, R. C., COURTENE-JONES, W., BOUCHER, J., PAHL, S., RAUBENHEIMER, K. & KOELMANS, A. A. 2024. Twenty years of microplastic pollution research-what have we learned? *Science,* 386**,** eadl2746.
THOMPSON, R. C., OLSEN, Y., MITCHELL, R. P., DAVIS, A., ROWLAND, S. J., JOHN, A. W. G., MCGONIGLE, D. & RUSSELL, A. E. 2004. Lost at Sea: Where Is All the Plastic? *Science*.
TING, W. H. T., TAN, I. A. W., SALLEH, S. F. & WAHAB, N. A. 2018. Application of water hyacinth (Eichhornia crassipes) for phytoremediation of ammoniacal nitrogen: A review. *Journal of Water Process Engineering,* 22**,** 239-249.
TISDALL, J. M. & OADES, J. M. 1982. Organic matter and water-stable aggregates in soils. *Journal of Soil Science,* 33.
TIWARI, N., SANTHIYA, D. & SHARMA, J. G. 2020. Microbial remediation of micro-nano plastics: Current knowledge and future trends. *Environ Pollut,* 265**,** 115044.
TOTSCHE, K. U., AMELUNG, W., GERZABEK, M. H., GUGGENBERGER, G., KLUMPP, E., KNIEF, C., LEHNDORFF, E., MIKUTTA, R., PETH, S., PRECHTEL, A., RAY, N. & KÖGEL-KNABNER, I. 2017. Microaggregates in soils. *Journal of Plant Nutrition and Soil Science,* 181**,** 104-136.
TOURNIER, V., DUQUESNE, S., GUILLAMOT, F., CRAMAIL, H., TATON, D., MARTY, A. & ANDRÉ, I. 2023. Enzymes' Power for Plastics Degradation - PubMed. *Chemical reviews,* 123.
TUN, T. Z., KUNISUE, T., TANABE, S., PRUDENTE, M., SUBRAMANIAN, A., SUDARYANTO, A., VIET, P. H. & NAKATA, H. 2022. Microplastics in dumping site soils from six Asian countries as a source of plastic additives. *Sci Total Environ,* 806**,** 150912.
TUNALI, M. M., MYRONYUK, O., TUNALI, M. & YENIGÜN, O. 2022. Microplastic Abundance in Human-Influenced Soils in Recreational, Residential, and Industrial Areas. *Water, Air, & Soil Pollution,* 233.
UNDAS, A. K., GROENEN, M., PETERS, R. J. B. & LEEUWEN, S. P. J. V. 2023. Safety of recycled plastics and textiles: Review on the detection, identification and safety assessment of contaminants. *Chemosphere,* 312.
UNEP 2021. Planetary action.
URBANUS, J. H., BRUNNER, A., BOERSMA, A., HENKE, S., KOOTER, I., LENSEN, S., PARKER, L., SCHWARZ, A., IMHOF, P., DORTMANS, A. & WIJNGAARD, M. 2022. Microplastics zijn overal: reductie met 70% haalbaar.
VAN CAUWENBERGHE, L. & JANSSEN, C. R. 2014. Microplastics in bivalves cultured for human consumption. *Environ Pollut,* 193**,** 65-70.
VAN DEN BERG, P., HUERTA-LWANGA, E., CORRADINI, F. & GEISSEN, V. 2020. Sewage sludge application as a vehicle for microplastics in eastern Spanish agricultural soils. *Environ Pollut,* 261**,** 114198.
VAN SCHOTHORST, B., BERIOT, N., HUERTA LWANGA, E. & GEISSEN, V. 2021. Sources of Light Density Microplastic Related to Two Agricultural Practices: The Use of Compost and Plastic Mulch. *Environments,* 8.
VAUGHAN, R., TURNER, S. D. & ROSE, N. L. 2017. Microplastics in the sediments of a UK urban lake. *Environ Pollut,* 229**,** 10-18.
VEJAN, P., KHADIRAN, T., ABDULLAH, R. & AHMAD, N. 2021. Controlled release fertilizer: A review on developments, applications and potential in agriculture. *J Control Release,* 339**,** 321-334.
VELEZ, J. F. M., SHASHOUA, Y., SYBERG, K. & KHAN, F. R. 2018. Considerations on the use of equilibrium models for the characterisation of HOC-microplastic interactions in vector studies. *Chemosphere,* 210**,** 359-365.
VERCAUTEREN, M., JANSSEN, C., ASSELMAN, J. & ZHANG, T. 2023a. Policy informing brief: Analysis on the link between microplastics, the environment and public health. *In:* UNIVERSITY, G. (ed.).
VERCAUTEREN, M., SEMMOURI, I., VAN ACKER, E., PEQUEUR, E., VAN ESCH, L., ULJEE, I., ASSELMAN, J. & JANSSEN, C. 2023b. Assessment of road run-off and domestic wastewater contribution to microplastic pollution in a densely populated area (Flanders, Belgium) - PubMed. *Environmental pollution,* 333.

VERSCHOOR, A., POORTER, L. D., DRÖGE, R., KUENEN, J. & VALK, E. D. 2016. Emission of microplastics and potential mitigation measures Abrasive cleaning agents, paints and tyre wear.
VERSCHOOR, A. J. & VAN DE VALK, E. 2018. Potential measures against microplastic emissions to water.
VIANELLO, A., BOLDRIN, A., GUERRIERO, P., MOSCHINO, V., RELLA, R., STURARO, A. & DA ROS, L. 2013. Microplastic particles in sediments of Lagoon of Venice, Italy: First observations on occurrence, spatial patterns and identification. *Estuarine, Coastal and Shelf Science,* 130**,** 54-61.
VIANELLO, A., JENSEN, R. L., LIU, L. & VOLLERTSEN, J. 2019. Simulating human exposure to indoor airborne microplastics using a Breathing Thermal Manikin. *Sci Rep,* 9**,** 8670.
VIAROLI, S., LANCIA, M. & RE, V. 2022. Microplastics contamination of groundwater: Current evidence and future perspectives. A review. *Sci Total Environ,* 824**,** 153851.
VOGEL, A., TENTSCHERT, J., PIETERS, R., BENNET, F., DIRVEN, H., VAN DEN BERG, A., LENSSEN, E., RIETDIJK, M., BROßELL, D., HAASE, A., VOGEL, A., TENTSCHERT, J., PIETERS, R., BENNET, F., DIRVEN, H., VAN DEN BERG, A., LENSSEN, E., RIETDIJK, M., BROßELL, D. & HAASE, A. 2024. Towards a risk assessment framework for micro- and nanoplastic particles for human health. *Particle and Fibre Toxicology 2024 21:1,* 21.
VOLLERTSEN, J. & HANSEN, A. A. 2016. Microplastic in Danish wastewater Sources, occurrences and fate.
VOM SAAL, F. & VANDENBERG, L. 2021. Update on the Health Effects of Bisphenol A: Overwhelming Evidence of Harm. *Endocrinology,* 162.
WAGNER, M., MONCLÚS, L., ARP, H. P. H., GROH, K. J., LØSETH, M. E., MUNCKE, J., WANG, Z., WOLF, R. & LISA, Z. 2024. State of the science on plastic chemicals - Identifying and addressing chemicals and polymers of concern.
WAHYUDI, A. A. J. & CORDOVA, M. R. 2016. Microplastic in the Deep-Sea Sediment of Southwestern Sumatran Waters. *Marine Research in Indonesia,* 41**,** 27-35.
WALKER, T. R. & FEQUET, L. 2023. Current trends of unsustainable plastic production and micro(nano)plastic pollution. *TrAC Trends in Analytical Chemistry,* 160.
WAN, Y., WU, C., XUE, Q. & HUI, X. 2019. Effects of plastic contamination on water evaporation and desiccation cracking in soil. *Science of The Total Environment,* 654.
WANG, J., LI, J., LIU, S., LI, H., CHEN, X., PENG, C., ZHANG, P. & LIU, X. 2021a. Distinct microplastic distributions in soils of different land-use types: A case study of Chinese farmlands. *Environ Pollut,* 269**,** 116199.
WANG, K., MIN, W., FLURY, M., GUNINA, A., LV, J., LI, Q. & JIANG, R. 2024. Impact of long-term conventional and biodegradable film mulching on microplastic abundance, soil structure and organic carbon in a cotton field. *Environ Pollut,* 356**,** 124367.
WANG, W., ZHANG, J., QIU, Z., CUI, Z., LI, N., LI, X., WANG, Y., ZHANG, H. & ZHAO, C. 2022. Effects of polyethylene microplastics on cell membranes: A combined study of experiments and molecular dynamics simulations. *J Hazard Mater,* 429**,** 128323.
WANG, Y., QIAN, H., WANG, Y. & QIAN, H. 2021b. Phthalates and Their Impacts on Human Health. *Healthcare,* 9**,** 603.
WANG, Y., WANG, F., XIANG, L., WANG, Z., SRIVASTAVA, P., JIANG, X. & XING, B. 2021c. Interaction between Differentially Charged Microplastics Particles and Soils. *SSRN Electronic Journal*.
WEBER, C. J., SANTOWSKI, A. & CHIFFLARD, P. 2022. Investigating the dispersal of macro- and microplastics on agricultural fields 30 years after sewage sludge application. *Sci Rep,* 12**,** 6401.
WHO 2022. Dietary and inhalation exposure to nano-and microplastic particles and potential implications for human health.
WILKENS, J. L., CALOMENI-ECK, A. J., BOYDA, J., KENNEDY, A. & MCQUEEN, A. D. 2024. Microplastic in Dredged Sediments: From Databases to Strategic Responses. *Bull Environ Contam Toxicol,* 112**,** 72.
WOODALL, L. C., GWINNETT, C., PACKER, M., THOMPSON, R. C., ROBINSON, L. F. & PATERSON, G. L. 2015. Using a forensic science approach to minimize environmental contamination and to identify microfibres in marine sediments. *Mar Pollut Bull,* 95**,** 40-6.
WU, J., WU, Z., YU, T., ZHANG, J., ZHANG, Z., WANG, H., ZHENG, Y., YANG, J. & WU, Y. 2024. Polyvinyl chloride and polybutylene adipate microplastics affect peanut and rhizobium symbiosis by interfering with multiple metabolic pathways. *J Hazard Mater,* 475**,** 134897.

XIA, F., YAO, Q., ZHANG, J. & WANG, D. 2021. Effects of seasonal variation and resuspension on microplastics in river sediments - PubMed. *Environmental pollution (Barking, Essex : 1987),* 286.

XU, J., ZUO, R., WU, G., LIU, J., LIU, J., HUANG, C. & WANG, Z. 2024. Global distribution, drivers, and potential hazards of microplastics in groundwater: A review. *Sci Total Environ,* 954**,** 176194.

XU, Z., ZHANG, Y., LIN, L., WANG, L., SUN, W., LIU, C., YU, G., YU, J., LV, Y., CHEN, J., CHEN, X., FU, L. & WANG, Y. 2022. Toxic effects of microplastics in plants depend more by their surface functional groups than just accumulation contents. *Sci Total Environ,* 833**,** 155097.

YA, H., JIANG, B., XING, Y., ZHANG, T., LV, M. & WANG, X. 2021. Recent advances on ecological effects of microplastics on soil environment. *Sci Total Environ,* 798**,** 149338.

YANG, J., LI, R., ZHOU, Q., LI, L., LI, Y., TU, C., ZHAO, X., XIONG, K., CHRISTIE, P. & LUO, Y. 2021a. Abundance and morphology of microplastics in an agricultural soil following long-term repeated application of pig manure. *Environ Pollut,* 272**,** 116028.

YANG, L., ZHANG, Y., KANG, S., WANG, Z. & WU, C. 2021b. Microplastics in soil: A review on methods, occurrence, sources, and potential risk. *Sci Total Environ,* 780**,** 146546.

YANSANEH, O. Y. & ZEIN, S. H. 2022. Latest Advances in Waste Plastic Pyrolytic Catalysis. *Processes,* 10.

YOOEUN, C. & YOUN-JOO, A. 2018. Current research trends on plastic pollution and ecological impacts on the soil ecosystem: A review. *Environmental Pollution,* 240.

YOON, J.-H., KIM, B.-H. & KIM, K.-H. 2024. Distribution of microplastics in soil by types of land use in metropolitan area of Seoul. *Applied Biological Chemistry,* 67.

YU, H., ZHANG, Y., TAN, W. & ZHANG, Z. 2022. Microplastics as an Emerging Environmental Pollutant in Agricultural Soils: Effects on Ecosystems and Human Health. *Frontiers in Environmental Science,* 10.

YU, L., ZHANG, J., LIU, Y., CHEN, L., TAO, S. & LIU, W. 2021. Distribution characteristics of microplastics in agricultural soils from the largest vegetable production base in China. *Sci Total Environ,* 756**,** 143860.

YU, Y. & FLURY, M. 2021. Current understanding of subsurface transport of micro- and nanoplastics in soil. *Vadose Zone Journal,* 20.

YUAN, J., MA, J., SUN, Y., ZHOU, T., ZHAO, Y. & YU, F. 2020. Microbial degradation and other environmental aspects of microplastics/plastics. *Sci Total Environ,* 715**,** 136968.

YUTAKA, M., SUCHANA, C., SANAE, C., REZA, C. M., ANDRÉS, C. C., FRANCOIS, G., PASCAL, H., HIROFUMI, H., ATSUHIKO, I., PETER, K., NIKOLAI, K., DAOJI, L., L., L. A., ELISA, M., A., M. S., JINGLI, M., HIROAKI, S., JOON, S. W., DHAMAR, S. A., HIDESHIGE, T., RICHARD, T., TADASHI, T., KEIICHI, U., KATERINA, V. & JUYING, W. 2023. Guidelines for Harmonizing Ocean Surface Microplastic monitoring methods.

ZAFIU, C., BINNER, E., BEIGL, P., VAY, B., EBMER, J. & HUBER-HUMER, M. 2023. The dynamics of macro- and microplastic quantity and size changes during the composting process. *Waste Management,* 162.

ZANTIS, L. J., KAZOUR, M., BORCHI, C., AGATI, R., COLPAERT, R., GIMBERT, F., VIJVER, M. G., PEIJNENBURG, W. & BOSKER, T. 2024. Quantitative tracking of nanoplastics along the food chain from lettuce (Lactuca sativa) to snails (Cantareus aspersus). *Science of The Total Environment,* 955.

ZHAI, Y., BAI, J., CHANG, P., LIU, Z., WANG, Y., LIU, G., CUI, B., PEIJNENBURG, W. & VIJVER, M. G. 2024. Microplastics in terrestrial ecosystem: Exploring the menace to the soil-plant-microbe interactions. *TrAC Trends in Analytical Chemistry,* 174.

ZHANG, G. S. & LIU, Y. F. 2018. The distribution of microplastics in soil aggregate fractions in southwestern China. *Sci Total Environ,* 642**,** 12-20.

ZHANG, G. S. & ZHANG, F. X. 2020. Variations in aggregate-associated organic carbon and polyester microfibers resulting from polyester microfibers addition in a clayey soil. *Environ Pollut,* 258**,** 113716.

ZHANG, G. S., ZHANG, F. X. & LI, X. T. 2019. Effects of polyester microfibers on soil physical properties: Perception from a field and a pot experiment. *Sci Total Environ,* 670**,** 1-7.

ZHANG, J., WANG, X., XUE, W., XU, L., DING, W., ZHAO, M., LIU, S., ZOU, G. & CHEN, Y. 2022. Microplastics pollution in soil increases dramatically with long-term application of organic composts in a wheat–maize rotation. *Journal of Cleaner Production,* 356.

ZHANG, L., HE, Y., JIANG, L., SHI, Y., HAO, L., HUANG, L., LYU, M. & WANG, S. 2024. Plastic additives as a new threat to the global environment: Research status, remediation strategies and perspectives. *Environ Res,* 263**,** 120007.

ZHANG, L., XIE, Y., LIU, J., ZHONG, S., QIAN, Y. & GAO, P. 2020a. An Overlooked Entry Pathway of Microplastics into Agricultural Soils from Application of Sludge-Based Fertilizers. *Environ Sci Technol,* 54**,** 4248-4255.
ZHANG, S., BAO, A., LIN, X., JIA, G. & ZHANG, Q. 2023a. Microplastic Accumulation in Agricultural Soils with Different Mulching Histories in Xinjiang, China. *Sustainability,* 15.
ZHANG, S., LIU, X., HAO, X., WANG, J. & ZHANG, Y. 2020b. Distribution of low-density microplastics in the mollisol farmlands of northeast China. *Sci Total Environ,* 708**,** 135091.
ZHANG, Y., YANG, S., ZENG, Y., CHEN, Y., LIU, H., YAN, X. & PU, S. 2023b. A new quantitative insight: Interaction of polyethylene microplastics with soil - microbiome - crop. *Journal of Hazardous Materials,* 460.
ZHOU, B., WANG, J., ZHANG, H., SHI, H., FEI, Y., HUANG, S., TONG, Y., WEN, D., LUO, Y. & BARCELÓ, D. 2020. Microplastics in agricultural soils on the coastal plain of Hangzhou Bay, east China: Multiple sources other than plastic mulching film. *Journal of Hazardous Materials,* 388.
ZHOU, Y., HE, G., JIANG, X., YAO, L., OUYANG, L., LIU, X., LIU, W. & LIU, Y. 2021. Microplastic contamination is ubiquitous in riparian soils and strongly related to elevation, precipitation and population density. *J Hazard Mater,* 411**,** 125178.
ZHOU, Y., LIU, X. & WANG, J. 2019. Characterization of microplastics and the association of heavy metals with microplastics in suburban soil of central China. *Sci Total Environ,* 694**,** 133798.
ZHOU, Z., HUA, J. & XUE, J. 2023. Polyethylene microplastic and soil nitrogen dynamics: Unraveling the links between functional genes, microbial communities, and transformation processes. *Journal of Hazardous Materials,* 458.
ZHU, J. & WANG, C. 2020. Recent advances in the analysis methodologies for microplastics in aquatic organisms: current knowledge and research challenges. *Anal Methods,* 12**,** 2944-2957.
ZIPFEL, C. & OLDROYD, G. E. 2017. Plant signalling in symbiosis and immunity. *Nature,* 543**,** 328-336.
ZWEIFEL, H. 2009. *Plastics additives handbook 6e*, Hanser Publishers.