Contents lists available at ScienceDirect

Science of the Total Environment

journal homepage: www.elsevier.com/locate/scitotenv

Review

Microplastics as an emerging threat to plant and soil health in agroecosystems

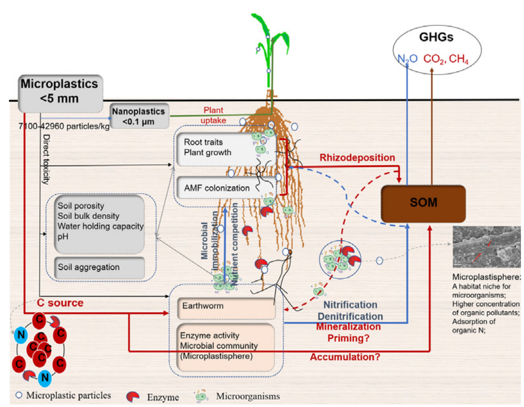
Jie Zhou [a], Yuan Wen [a], Miles R. Marshall [b], Jie Zhao [a], Heng Gui [c,d], Yadong Yang [a], Zhaohai Zeng [a], Davey L. Jones [b,e], Huadong Zang [a,*]

[a] College of Agronomy and Biotechnology, China Agricultural University, Beijing, China
[b] School of Natural Sciences, Bangor University, Bangor, Gwynedd LL57 2UW, UK
[c] CAS Key Laboratory for Plant Diversity and Biogeography of East Asia, Kunming Institute of Botany, Chinese Academy of Science, Kunming, China
[d] Centre for Mountain Futures (CMF), Kunming Institute of Botany, Chinese Academy of Science, Kunming, Yunnan, China
[e] Soils West, UWA School of Agriculture and Environment, The University of Western Australia, Perth, WA 6009, Australia

HIGHLIGHTS

- MPs can alter rhizodeposition input and soil organic matter decomposition.
- Effects on soil C and nutrient cycling depend on MP types, concentration, size, and shape.
- Nano-sized MPs can accumulate in roots and be transported to the shoot.
- Bio-based MPs can exert strong negative effects on plant by increasing nutrient competition.

GRAPHICAL ABSTRACT



ARTICLE INFO

Article history:
Received 14 January 2021
Received in revised form 22 April 2021
Accepted 26 April 2021
Available online 30 April 2021

Editor: Thomas Kevin V

Keywords:
Microplastics
Plant growth
Soil carbon storage
Nutrient cycling
Greenhouse gas emissions
Biodegradable plastics
Agroecosystem

ABSTRACT

Microplastics (MPs, <5 mm in diameter) have been widely recognized as a critical environmental issue due to their extensive use and low degradation rate. Based on current evidence, our aim is to evaluate whether MPs represent an emerging threat to plant-soil health in agroecosystems. We assess the ecological risks to plant-microbe-soil interactions associated with MPs and discuss the consequences of MPs on soil carbon (C), nutrient cycling, as well as greenhouse gas emissions in agroecosystems. We also identify knowledge gaps and give suggestions for future research. We conclude that MPs can alter a range of key soil biogeochemical processes by changing its properties, forming specific microbial hotspots, resulting in multiple effects on microbial activities and functions. Mixed effects of MPs on plant growth and performance can be explained by the direct toxicity of MPs or the indirect alteration in soil physical structures and microbial communities (i.e. symbiotic arbuscular mycorrhizal fungi). Because of the diverse nature of MPs found in soils, in terms of polymer type, shape and size, we also see differing effects on soil organic matter (SOM) decomposition, nutrient cycling, and greenhouse gases production. Importantly, increased bioavailable C from the decomposition of biodegradable MPs, which enhances microbial and enzymatic activities, potentially accelerates SOM mineralization and increases nutrient competition between plant and microbes. Thus, biodegradable MPs appear to pose a greater risk to plant growth compared to petroleum-based MPs. Although MPs may confer some benefits in agroecosystems (e.g. enhanced soil structure, aeration), it is thought that these will be far outweighed by the potential disbenefits.

* Corresponding author.
E-mail address: zanghuadong@cau.edu.cn (H. Zang).

https://doi.org/10.1016/j.scitotenv.2021.147444
0048-9697/

Contents

1. Introduction . . . 2
2. Effect of MPs on soil health . . . 3
 2.1. Effects of MPs on soil properties . . . 3
 2.2. Effects of MPs on soil microbial community and function . . . 3
 2.3. Effects of MPs on soil fauna . . . 5
3. Effect of MPs on plant health . . . 6
 3.1. Direct effect of MPs on plant growth . . . 6
 3.2. Indirect effect of MPs on plant growth . . . 7
 3.3. Uptake of MPs by plants . . . 7
4. Effects of MPs on soil C and nutrient cycling . . . 8
 4.1. Effect of MPs on belowground C inputs . . . 8
 4.2. Effect of MPs on SOM decomposition . . . 9
 4.3. Effect of MPs on nutrient cycling and GHGs emission . . . 9
5. Conclusions and future prospects . . . 9
Declaration of competing interest . . . 10
Acknowledgments . . . 10
References . . . 10

1. Introduction

Synthetic polymers are widely used in our daily lives with more than 359 million tons of plastics produced annually (Plastics Europe, 2019). It is predicted that by 2050 plastic waste production will reach 12,000 million tons (Geyer et al., 2017). Although a large proportion of plastic is recycled, repurposed or incinerated, 32% of all plastic waste still finds its way into the natural environment (i.e. terrestrial or aquatic ecosystems) (Geyer et al., 2017; de Souza Machado et al., 2018a). Approximately 80% of marine plastic waste is terrestrially derived and it is in the terrestrial biosphere where we often see the highest concentrations (Andrady, 2011; Jambeck et al., 2015). Thus, plastic accumulation in terrestrial ecosystems represents a long term plastic reservoir which may impact on freshwater and marine ecosystems for decades to come.

An important source of plastics in terrestrial ecosystems is agroplastics (plastics used in agriculture), whose global usage exceeded 6 million tons in 2018 (Fig. 1b; Sintim and Flury, 2017). It is estimated that only 6–26% of plastic debris is recycled (Fig. 1c; Alimi et al., 2018), with the remainder becoming fragmented into microplastics (MPs, <5 mm) through physical abrasion, UV irradiation, thermal oxidation, and microbial processing (Rillig et al., 2017a, b; He et al., 2018). MPs can be introduced into agroecosystems via multiple pathways including fertilizer coatings (Heuchan et al., 2019), wastewater irrigation (Zhang and Liu, 2018), compost addition, biosolids application (Nizzetto et al., 2016; Weithmann et al., 2018), and importantly the use of mulching film (Liu et al., 2014; Qi et al., 2018). In general, MPs degradation in soils is extremely slow, typically taking hundreds and possibly thousands of years for full mineralization to occur (Zubris and Richards, 2005; Andrady, 2011). In many countries, plastic particles are more abundant in agricultural soils rather than in urban soils because of frequent plasticulture usage (5–35 kg plastic film ha^{-1} yr^{-1}; Liu et al., 2014) and slower rates of overland flow (Nizzetto et al., 2016). In China, where the use of plastics in agroecosystems is widespread, concentrations in soils typically range between 7100 and 42,900 plastic particles kg^{-1} soil (mean 18,760 particles kg^{-1} soil), 95% of which are between 0.05 and 1 mm in size (i.e. MPs) (Zhang and Liu, 2018). With such high concentrations found in agroecosystems it is vitally important to evaluate the impacts of MPs on plant-soil health.

Once in the soil, MPs can either directly or indirectly impact ecosystem functions and plant-soil health. Chemical additives (i.e. plasticizers) used in the manufacture of MPs may be toxic to soil organisms (Prata et al., 2020). Due to the chemical inertia and structural characteristics, MPs can also sorb toxic organic and inorganic compounds (e.g., heavy metals, xenobiotics and pathogens) from the surrounding soils (Hahladakis et al., 2018; Wang et al., 2020), which may contribute to a greater ecological risk in terms of plant-soil health (Huerta Lwanga et al., 2017; Li et al., 2020). MPs can alter soil physico-chemical properties, i.e. pH, soil aggregation, bulk density, and water holding capacity (de Souza Machado et al., 2018a, b, 2019; Wan et al., 2019), which in turn have a diverse range of effects on microbial functions as well as plant growth and performance (as listed in the Tables 1 and 2). Given that microorganisms constitute the main biological population in soils and play a vital role in biogeochemical cycling (Kuzyakov and Xu, 2013), the interactions between MPs and soil microbes may result in an unpredictable consequence on plant and soil health. Due to the high degree of variability in polymer type, size, shape and concentration, the impacts of MPs on soil biogeochemical processes and its underlying mechanisms still remain unclear. Considering the important role of soil organic matter (SOM) in maintaining soil fertilizer, ecosystem stability, nutrient cycling, as well as crop yields (Lehmann and Kleber, 2015), the potential effects of MPs on soil C and nutrient cycling in agroecosystems remain relatively unexplored and should be a research priority. It is therefore necessary to critically evaluate the current evidence based on MPs behaviour and fate in soils and to identify key knowledge gaps and research priorities.

Increasing concerns surrounding MPs pollution in agroecosystems have led to the development of biodegradable polymers in an attempt to decrease the use of petroleum-based plastics (Volova et al., 2017). Unlike petroleum-based MPs, biodegradable MPs can be broken down relatively quickly by a range of organisms and are not thought to produce any harmful by-products (Volova et al., 2017; Sander, 2019). Comparatively, however, very little is known about the impacts of biodegradable MPs on plant-soil interactions, despite their increasing use in many countries.

Overall, MPs have become a global environmental issue and have aroused widespread concern about their potential ecological risks. Due to our poor understanding of plastic behaviour in soils, it is currently not possible to make informed decisions on future policies relating to the safe use in agroecosystems. It is therefore essential to systematically investigate the safety of MPs in agroecosystems, particularly plastic mulching film, whose usage is widespread globally. In this study, we searched the databases of Web of Science (WOS) for studies published between 1991 and August 2020 with the keywords of "microplastics" in conjunction with "agroecosystem". In total, 159 research articles were found with most published between 2016 and 2020 (Fig. 2). Among these publications, 49% and 26% were produced by corresponding authors in China and the USA, respectively. The top three sources which contained the most publications about MPs in agroecosystems were: Science of the Total Environment (35%), Environment Pollution (20%), and Environmental Science and Technology (11%) (Fig. 2c).

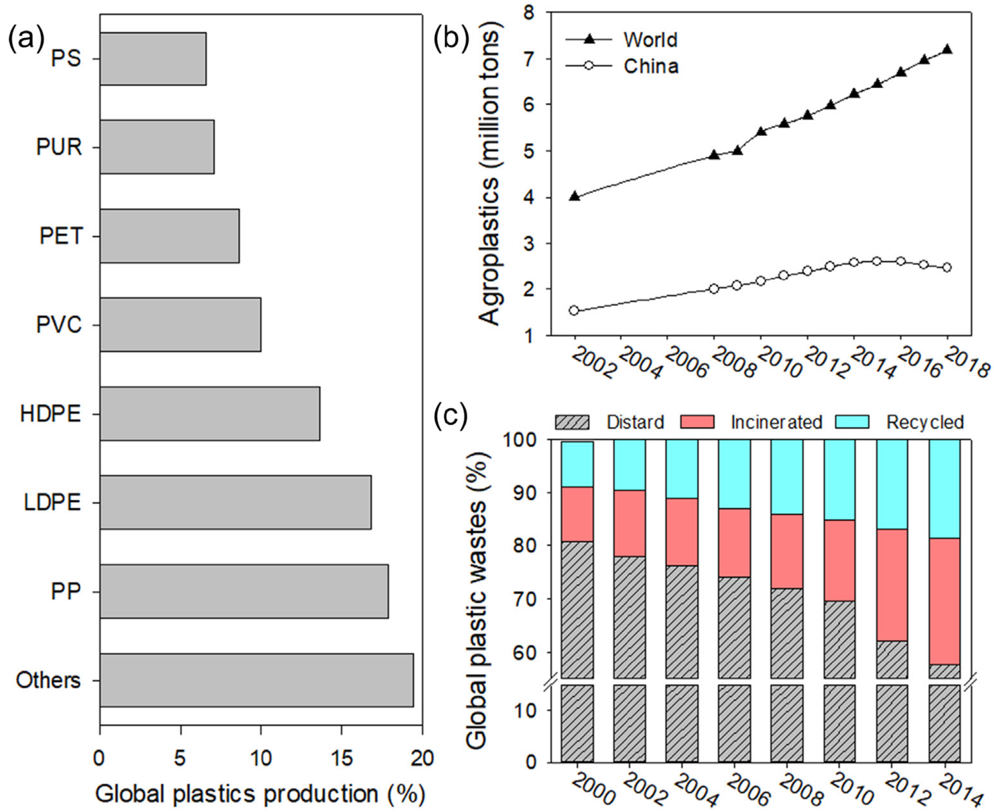


Fig. 1. The common types of plastics used in 2015 globally (a), total agroplastic production in the world and China (b), as well as the global percentage of discarded, incinerated, and recycled plastics from 2000 to 2014 (c). Agroplastics production accounts for 2% of the total plastic production worldwide (Geyer et al., 2017). The data of agroplastics production in China was extracted from the National Bureau of statistics-China NBSC (2018). The data about the percentage of global plastics waste recycled, incinerated, and discarded as well as the types of plastics were obtained from Plastics Europe (2019). LDPE, low-density polyethylene; HDPE, high-density polyethylene; PVC, polyvinyl chloride; PP, polypropylene; PS, polystyrene; PET, polyethylene terephthalata; PUS, polyarylsulfone.

Here, we aim to (1) study how MPs impact soil physical structure, microbial activity and community, and soil fauna; (2) investigate whether MPs are an emerging threat to plant health (both directly and indirectly); (3) estimate the effect of MPs on C storage and balance (i.e. C input and SOM decomposition), nutrient cycling and greenhouse gases emissions in agroecosystems; (4) estimate whether biodegradable MPs pose a potential risk to plant-soil health; and (5) identify future challenges in related areas.

2. Effect of MPs on soil health

2.1. Effects of MPs on soil properties

Due to their distinct characteristics, MPs can influence soil properties by changing its physical structures (Fig. 3). Firstly, MPs may alter soil aggregation due to their binding to soil mineral and organic components (Rillig et al., 2017a; Lei et al., 2018), but the effect is expected to vary with composition, size, type and shape of the plastic particles. For example, the presence of polyester fibers increased water-stable aggregates (de Souza Machado et al., 2018b), while polyethylene film decreased the proportion of large macroaggregates (>2 mm) (Zhang et al., 2019). Since microbial metabolites (i.e. exo-biopolymers like polysaccharides) function as gluing substances, promoting soil stability (Lehmann et al., 2017), the desorption of toxic additives (i.e. phthalate esters) used in the manufacture of plastics can affect microbial activity, subsequently impacting soil aggregation. *Actinobacteria*, one of the most important bacterial groups contributing to soil aggregation (Lehmann et al., 2017), have been shown to reduce in abundance and richness due to the presence of microplastic films in soils (Huang et al., 2019; Fei et al., 2020). Secondly, because plastics are often less dense than many soil minerals, MPs can decrease soil bulk density (de Souza Machado et al., 2018b), increasing soil aeration which may aid root penetration. Thirdly, polyethylene has been shown to lower soil pH (Boots et al., 2019), whilst polylactic acid may increase soil pH (Qi et al., 2020). Any MPs changes in soil pH are likely to have a large influence on soil microbial community structures and activities which are highly responsive to pH (Rousk et al., 2009; Jones et al., 2019). In terms of soil water dynamics, polyester fibers (8 μm) enhanced water holding capacity potentially keeping soils saturated for longer periods (de Souza Machado et al., 2019), while polyethylene films (2 mm) increased soil water loss through increased evaporation (Wan et al., 2019). The former was due to the ability of fibers to form soil clumps and entangle soil particles at finer spatial scales. However, MPs with a larger size may negatively affect soil water holding capacity and induce anoxia (Liu et al., 2014). Alterations in soil water content could alleviate or aggravate drought, which is predicted to increase as a result of climate warming over the next few decades (Lozano and Rillig, 2020). Therefore, changes in soil structures causing alterations in soil hydrological dynamics induced by MPs with different concentrations, types, sizes and shapes may result in unpredictable impacts under future climate change scenarios.

2.2. Effects of MPs on soil microbial community and function

Soil microorganisms are important players in biogeochemical cycling, which are the basis for food production and climate regulation. Therefore, understanding the response of soil microorganisms to MPs will allow us to predict potential ecosystem-level outcomes resulting from MPs pollution. MPs could affect soil properties (as described in Section 2.1), change biophysical environments, and substantially

Table 1
The effects of microplastics on soil properties, microbial activities, and functions in the agroecosystem depending on the type, shape, size, and concentration of polymer plastics.

Polymers	Shape	Size (μm)	Concentration (%)	Soil structure and microbial biomass/activity/species	Effects[b]	References
Polyethylene (PE)[a]	Powder	125	1, 5, 10, 20	β-1,4-Glucosidase	n	Zang et al., 2020
				Xylosidase	n	
				Cellobiohydrolase	n	
				Chitinase	n	
				Leucine aminopeptidase	n	
				Microbial biomass	+	
	Powder	<150	7, 28, 45, 60	Earthworm (*Lumbricus terrestris*)	−	Huerta Lwanga et al., 2017
	Fragment	643	0.05, 0.1, 0.2, 0.4, 1, 2	Bulk density	−	de Souza Machado et al., 2018a, b
				Fluorescein diacetate hydrolase	n	
				Microbial biomass	n	
	Film	678	1, 5	Fluorescein diacetate hydrolase	−	Fei et al., 2020
				Bacterial family associated with nitrogen fixation	+	
	Fragment	>800	2	Microbial biomass	+	de Souza Machado et al., 2019
	Fragment	50–1000	0.5, 1, 2	pH	+	Qi et al., 2020
				Soil C:N	+	
				Electrical conductivity	−	
				Porosity	−	
	Fragment	<2000	1	Earthworm (*Eisenia fetida*)	n	Judy et al., 2019
				Urease	+	
				Acid phosphatase	+	
	film	5000	ns	*Acidobacteria*, *Bacteriodietes*	+	Huang et al., 2019
Polyvinyl chloride (PVC)	ns[c]	18	1, 5	Fluorescein diacetate	−	Fei et al., 2020
				Hydrolase	+	
				Urease	+	
				Sphingomonadaceae	−	
	Powder	80–250	0.1	Collembolan (*Folsomia candida*)	−	Zhu et al., 2018
	Powder	125	1, 5, 10, 20	β-1,4-Glucosidase	−	Zang et al., 2020
				Xylosidase	−	
				Cellobiohydrolase	n	
				Chitinase	n	
				Leucine aminopeptidase	n	
				Microbial biomass	+	
	Fragment	<2000	1	Earthworm (*Eisenia fetida*)	n	Judy et al., 2019
Polyester (PES)	Fiber	5	0.1, 0.3	Bulk density	n	Zhang et al., 2019
				Saturated hydraulic conductivity	n	
	Fiber	8	0.05, 0.1, 0.2, 0.4, 1, 2	Bulk density	n	de Souza Machado et al., 2018a, b
				Water holding capacity	+	
				Microbial biomass	−	
	Fiber	40	0.1, 1	Earthworm (*Lumbricus terrestris*)	n	Prendergast-Miller, 2019
	Fiber	8	0.2	Microbial biomass	+	de Souza Machado et al., 2019
				Arbuscular mycorrhizae	+	
Polyethylene terephthalate (PET)	Fragment	222–258	2	Arbuscular mycorrhizae	−	de Souza Machado et al., 2019
				Microbial biomass	−	
	Fragment	<2000	1	Earthworm (*Eisenia fetida*)	n	Judy et al., 2019
Polypropylene (PP)	Powder	180	7, 28	Phenol oxidase	+	Liu et al., 2017
				Fluorescein diacetate hydrolase	+	
	Fragment	647–754	2	Arbuscular mycorrhizae	+	de Souza Machado et al., 2019
				Microbial biomass	+	
Biodegradable MPs						
Polyacrylic (PLA)	Fiber	18	0.05, 0.1, 0.2, 0.4, 1, 2	Bulk density	−	de Souza Machado et al., 2018a, b
				Water holding capacity	n	
				Fluorescein diacetate	+	
				Hydrolase	−	
				Microbial biomass	−	
	Fiber	15–20	2	Microbial biomass	+	de Souza Machado et al., 2019
	ns	20–50	2	β-Glucosidase	n	Chen et al., 2020
				Urease	n	
				Catalase	n	
				Microbial biomass	n	
Poly(3-hydroxybutyrate-*co*-3-hydroxyvalerate) (PHBV)	Powder	125	10	β-1,4-Glucosidase	+	
				Leucine aminopeptidase	+	
				Acid phosphatase	+	
				Microbial biomass	+	
				Acidbacteria, *Bacteroidetes*	+	Zhou et al., 2021
Starch-based biodegradable plastic (Bio)	Fragment	50–1000	1	pH	+	Qi et al., 2020
				Soil C:N	+	
				Electrical conductivity	−	
				Bacillus, *Variovorax*	+	

[a] PE includes both low and high density polyethylene.
[b] +: positive; −: negative; n: neutral. Note, the 'positive' and 'negative' effects mean the significant difference between with and without MPs at $p < 0.05$ level.
[c] ns: not mentioned in the publication.

Table 2
The effects of microplastics on plant health in the agroecosystem depending on the type, shape, size, and concentration of polymer plastics.

Polymers	Shape	Size (μm)	Concentration (%)	Crop	Effects[b]	References
Polyethylene terephthalate (PET)	Fiber	222–258	2	Spring onion	n	de Souza Machado et al., 2019
		<2000	1	Wheat	n	Judy et al., 2019
Polyvinyl chloride (PVC)	Powder	125	1	Wheat	–	Zang et al., 2020
			5		–	
			10		+	
			20		+	
	Fragment	<2000	1	Wheat	n	Judy et al., 2019
Polyethylene (PE)[a]	Powder	125	1	Wheat	–	Zang et al., 2020
			5		–	
			10		+	
			20		+	
	Fragment	50–1000	1	Wheat	–	Qi et al., 2018
	ns	100–154	0.1	Maize	n	Wang et al., 2020
			1		n	
			10		n	
	Fragment	>800	2	Spring onion	n	de Souza Machado et al., 2019
Polyester (PES)	Fiber	5	0.2	Spring onion	+	de Souza Machado et al., 2019
Polypropylene (PP)	Fragment	647–754	2	Spring onion	n	de Souza Machado et al., 2019
Bioplastics						
Polyacrylic (PLA)	Fragment	15–20	2	Spring onion	–	de Souza Machado et al., 2019
	ns	100–154	0.1	Maize	n	Wang et al., 2020
			1		n	
			10		–	
Poly(3-hydroxybutyrate-co-3-hydroxyvalerate) (PHBV)	Powder	125	10	Wheat	–	Zhou et al., 2021
Starch-based biodegradable plastic (Bio)	Fragment	50–1000	1	Wheat	–	Qi et al., 2018

[a] PE includes both low and high density polyethylene.
[b] Effects indicated MPs on plant growth or biomass; +: stimulate plant growth; −: inhibit plant growth; n: no response.

influence soil microbial communities and functioning (Rillig et al., 2019b). MPs could serve as a novel ecological habitat for microorganisms living at the soil-plastic interface (i.e. microplastisphere), allowing the formation of unique microbial communities (Zhou et al., 2021). We hypothesize that MPs may attract or favor specific microbial taxa, and interfere with belowground plant-microbes interactions, forming microbial hotspots in the microplastisphere (Zang et al., 2020; Zhou et al., 2020a, b). For example, polyethylene fragments induced abundant taxa including plastic-degrading bacteria and pathogens (Huang et al., 2019). Biodegradable MPs (PHBV) increased the abundance of oligotrophic microorganisms and decreased the fast-growing copiotrophs (Zhou et al., 2021). In summary, MPs can provide novel microbial niches promoting the proliferation of specific microbial groups which may result in unpredictable consequences on ecosystem functions.

Besides forming microbial hotspots, MPs can have divergent influences on soil microbial communities and enzyme activities (Table 1), e.g., activation (Liu et al., 2017; de Souza Machado et al., 2019), suppression (Fei et al., 2020), or remaining unchanged (Zang et al., 2020). For example, polyacrylic and polyester fibers (0.1%) decreased microbial metabolic activity (Judy et al., 2019), while polyethylene fragments (0.4%) had little effect on soil microbial diversities (de Souza Machado et al., 2018b). Polyvinyl chloride and polyethylene powders (10%) in a wheat-soil system resulted in a microbial community shift from Gram-positive to Gram-negative bacteria, and decreased β-glucosidase and xylosidase activities by 16–43% (Zang et al., 2020). Polyethylene film increased the microbial abundance of *Acidobacteria*, *Bacteriodietes*, *Gemmatimonadetes*, and *Nitrospirae* (Huang et al., 2019), while polyvinyl chloride film (1–5%) significantly reduced the abundance of the family *Sphingomonadaceae* (Fei et al., 2020). The observed inconsistent results on microbial activities and community structures could be explained by the varied chemical makeup, specific surface area, and hydrophobicity of MPs, as well as the modified soil structures (e.g., pH, soil aggregate stability, porosity, and water content) (Fig. 3; Rillig et al., 2019b; Yu et al., 2020; Seeley et al., 2020). Polymer type was a key MPs variable explaining the variance in microbial activity, with polypropylene fragments and polyethylene films were the polymers that decreased microbial activity the most (Lozano et al., 2021b). Due to the variations in sorption capacities, the shape of different MPs provides specific microbial habitats. MPs with different sizes affect microbial communities differently due to the variations in surface-to-volume ratios of MPs particles (Brodhagen et al., 2017). Specifically, nanoplastics (<0.1 μm) may be able to penetrate cell membranes and thus exert cytotoxic effects (Lei et al., 2018), because of the bioaccumulation in the cells of yeasts and filamentous fungi (Miyazaki et al., 2015; Nomura et al., 2016; de Souza Machado et al., 2018a). This highlights the potential for nanoplastics to enter and accumulate in the soil-detritus food web to cause biological effects on microbes, whereas altered soil properties caused by nanoplastics may be less important. Given that water-stable aggregates are considered important habitats for soil microorganisms and hotspots of microbial processes, a decrease in water-stable aggregates caused by polyester fibers may result in significant impacts to plant-soil health (de Souza Machado et al., 2020). However, a mechanistic understanding of the interaction between MPs and microbial community still remains unknown and is a critical knowledge gap that needs to be filled in order to better predict the ecological consequences of MPs in soils.

2.3. Effects of MPs on soil fauna

Soil fauna plays an important role in ecosystem functions, especially earthworms whose abundance is viewed as a key biological indicator when assessing soil quality (Bünemann et al., 2018). Low levels of microplastic (0–0.5%) do not affect earthworm growth and survival (Huerta Lwanga et al., 2016), but high concentrations (1–2%) suppressed earthworm growth and thus increased their mortality (Cao et al., 2017). This indicated that MPs may have a direct toxic effect on soil fauna (Table 1; Fig. 3), which was dependent on the concentration of MPs. The ingestion of MPs is also the key factor controlling its toxicity to soil fauna, and its potential to bioaccumulate in the food chain (Rillig et al., 2017b; Zhu et al., 2018). In addition, MPs can adsorb and concentrate hazardous chemicals on their surface, further increasing the risk posed to organisms and humans (Hahladakis et al., 2018; Wang et al., 2020). A mesocosm study showed that polyethylene fragments could serve as vectors increasing the bioavailability of zinc to earthworms (Hodson et al., 2017). When MPs fragment further into smaller particles (<1 μm), potential hazards might become more concerning due to the

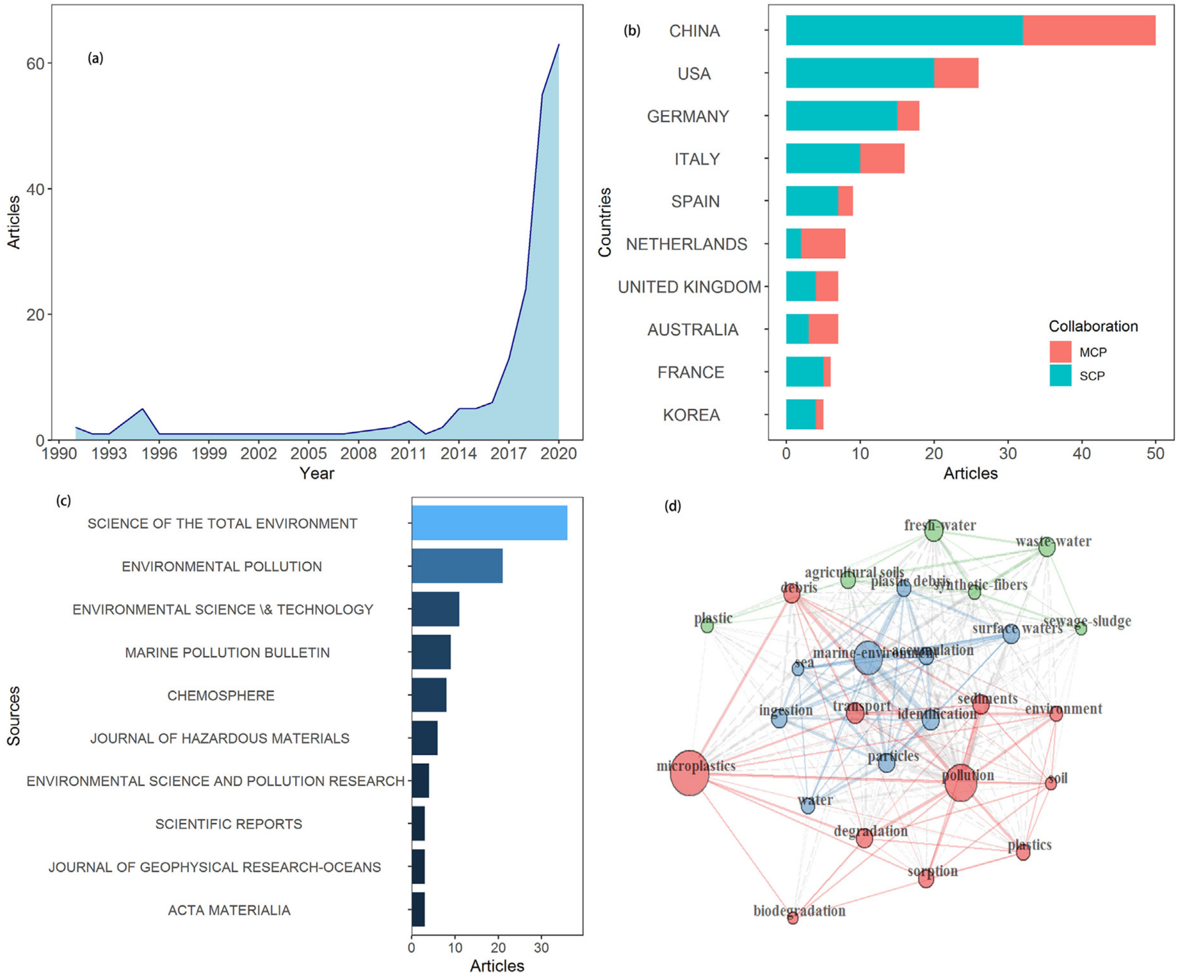


Fig. 2. The number of microplastic publications in agroecosystems per year from 1991 to 2020 (a), the source of publications based on corresponding author (b), the journals of publications (c), and hotspots in microplastic research: map of microplastic topics based on keywords constructed using R software (d). The size of each circle represents the frequency of that keyword. All graphs were produced based on the ISI Web of Science (WOS) database for the following combinations of terms within a date range of 1991 to 2020: microplastic + agroecosystem.

poorly soluble biopersistent (Prata et al., 2020). Jeong et al. (2016) found that the smaller polystyrene powders (0.05 μm) had a greater toxic influence relative to the larger MPs (0.5 and 6 μm) on rotifer species. Moreover, MPs could indirectly affect the biological behaviour of soil fauna by alternating soil structures as discussed in Section 2.1. In turn, earthworms also act as horizontal and vertical transport vectors of MPs, incorporating MPs more widely into the soil via their casts, burrows, and adherence to the earthworm's exterior, thus leading to an increased risk of exposure for other soil organisms (Rillig et al., 2017b). Overall, although the effect of MPs pollution on soil fauna has received great attention, research has almost exclusively focused on earthworms, the interaction between MPs, soil fauna, and other soil biota requires further investigation.

3. Effect of MPs on plant health

3.1. Direct effect of MPs on plant growth

There is increasing evidence showing that MPs can affect plant growth and performance as listed in Table 2 (Qi et al., 2018; de Souza Machado et al., 2019; Zang et al., 2020; Zhou et al., 2021). The biggest impact of MPs on plants is in their roots, followed by leaves, shoots and then stems. This is because MPs are easily absorbed by roots from contaminated soils and by atmospheric deposition to above ground plant parts (Zhang et al., 2020). MPs can cause a delay in germination (Bosker et al., 2019), inhibit both above- and below-ground growth of wheat in both vegetative and reproductive stages (Qi et al., 2018), and elicit toxicity to *Vicia faba* (Jiang et al., 2019). The phytotoxic effect could be attributed to the presence of additives (i.e. plasticizers and flame retardants) incorporated into plastics (Hahladakis et al., 2018; Wang et al., 2020) or other secondary pollutants (i.e. antibiotics and heavy metals) adsorbed onto their surface (Fig. 3; Shen et al., 2019; Wang et al., 2020). These chemical additives may be weakly bound, or not bound at all to the polymer molecule, and thus easily leach into the soil, resulting in adverse effects on plant growth (Hahladakis et al., 2018; Bolan et al., 2020). It is generally believed that toxicity increases with increasing adsorption capacity of MPs (Du et al., 2020), which depends on the type, size, shape of MPs (Wang et al., 2016a). For instance, polyamide had a greater affinity sorbing antibiotics than polyvinyl chloride, polyethylene, and polypropylene, due to its porous structure and hydrogen bonding between its amide group (proton donor group) and the antibiotic's carbonyl groups (proton acceptor group) (Li et al., 2018). Additionally, toxic effects on plants were influenced by MPs size, and the smaller the particle size, the greater the harm to plants (Li et al., 2020). For example, nanoplastics can slow down or completely inhibit water and nutrient uptake by adhering to the surface of seeds physically blocking pores (Bosker et al., 2019). Overall, MPs can act as vectors and sinks of toxic pollutants in their surroundings, causing phytotoxicity and directly inhibit plant growth.

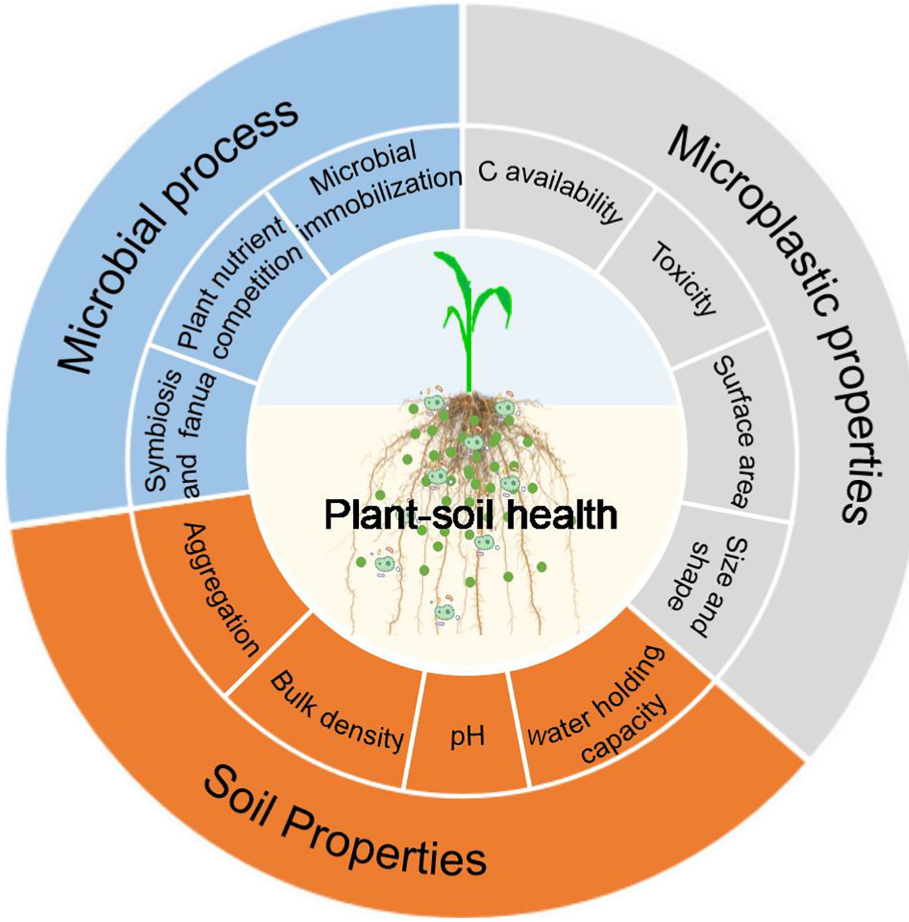


Fig. 3. Schematic overview of the connection between primary microplastics properties, the soil processes they may influence, and the plant-microbe-soil health. Microplastics toxicity, surface area, shape, and size all influence soil microbial processes, symbiosis, and soil fauna, therefore, the soil organic matter (SOM) decomposition and plant growth. Microplastics C availability can also be a determinant due to its ability to stimulate specific microbial groups, as well as cause microbial nutrient immobilization, thus induce a potential risk on plant-soil health. Microplastics size and shape could also alter soil properties, i.e. may change soil aggregation stability, pH, bulk density, and water holding capacity, thus cause a diverse effect on microbial functions as well as plant performance and growth.

3.2. Indirect effect of MPs on plant growth

MPs can be defined as a soil physical contaminant which may indirectly affect plant growth (see Section 2.1; Fig. 3). For instance, MPs fibers lowered soil bulk density and enhanced soil aeration (de Souza Machado et al., 2018b), which can reduce root penetration resistance and increase root growth (Rillig et al., 2019a). Likewise, the positive effect of MPs on shoots can also be linked to the reduction of soil bulk density (Lozano and Rillig, 2020). By contrast, increased earthworm mortality caused by polyethylene powder resulted in an indirect effect on soil porosity and water content (Huerta Lwanga et al., 2017), potentially suppressing plant growth. Changes in soil structure could also influence microbial composition and functions (de Souza Machado et al., 2019), which may affect soil fertility and a range of rhizosphere processes (Qi et al., 2020). Specifically, soil microbes such as symbiotic arbuscular mycorrhizal fungi (AMF) have a direct effect on plant growth (Fig. 4). To date, only one study by de Souza Machado et al. (2019) has shown that AMF colonization increases in the presence of polyester and polypropylene but decreases in the presence of polyethylene terephthalate. Therefore, MPs-induced alterations in soil and microbial properties may result in a diverse range of indirect effects on plant health, and detailed research needs to be conducted to clarify the underlying mechanisms. This should also include other studies on other key symbionts of agricultural importance (e.g., AMF and N-fixing communities).

Increased plant nutrient stress can result from the high C:N ratio resulting in microbial N immobilization from the addition of MPs (Fig. 3; Volova et al., 2017; Boots et al., 2019; Rillig et al., 2019a). The negative impact on plant growth will be potentially greatest when non-petroleum based biodegradable MPs are used due to their greater bioavailability (Qi et al., 2018, 2020; Zhou et al., 2021). For instance, a biodegradable starch based MPs had a greater negative impact on wheat height and biomass compared to a non-degradable petroleum-based MPs (Qi et al., 2018). Furthermore, biodegradable MPs (i.e. PHBV) caused wheat death during a 4-weeks study (Zhou et al., 2021), which may be attributed to the intermediate and/or final metabolites produced during PHBV degradation. Overall, although biodegradable MPs have been heralded as a sustainable alternative to petroleum-based plastics, our review indicates that it is also important to consider the potential disbenefits of such material on plant growth and performance.

3.3. Uptake of MPs by plants

Due to their high molecular weight and large size preventing their penetration through cellulose-rich plant cell walls, it is not expected that plants are able to take up MPs (Teuten et al., 2009). However, when MPs break down to nanoparticles (<0.1 μm), they can traverse biological membranes and enter plant cells, potentially entering the food chain (Jassby et al., 2019; Li et al., 2020). This was demonstrated by

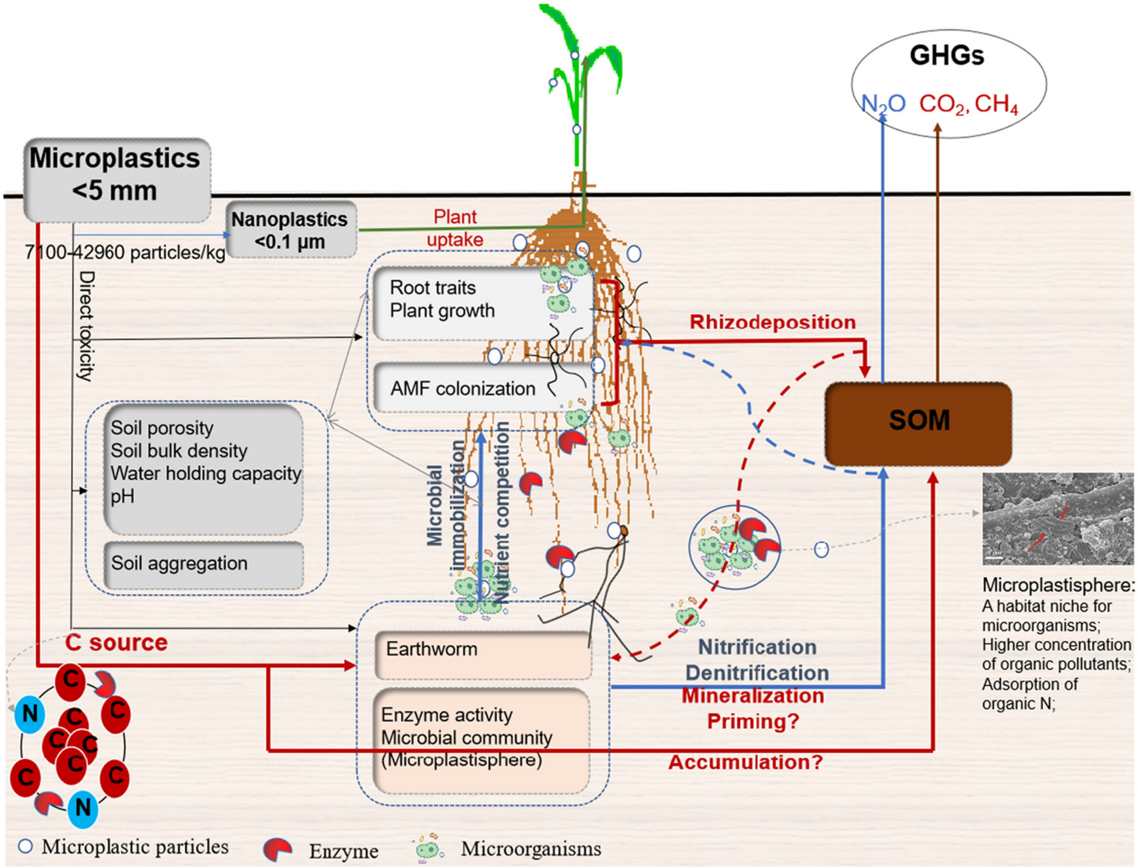


Fig. 4. Graphical abstract about the effects of microplastics on plant-soil health, and soil organic matter (SOM) decomposition, as well as interactions with soil microorganisms and plants. Toxicity due to the additives and pollutants adsorbed on the surface of microplastic could cause a direct inhibition on plant growth, arbuscular mycorrhizal fungi symbiosis, as well as soil fauna and microbial groups. Microplastics could alter soil properties, therefore influence microbial activity and plant growth. Further, the C in the microplastics may accumulated in the soil, or prime the microbial community to mineralize native SOM, thus cause greenhouse gases (CO_2, N_2O, CH_4) emissions. Although the mechanisms underlying some of these factors are still elusive and interactions among factors are not well understood, microplastics could pose a potential risk to soil and plant health in agroecosystems.

Bandmann et al. (2012) who found that tobacco BY-2 cells could take up nano-polystyrene (0.02 and 0.04 μm) in cell culture. Similarly, polystyrene nanoparticles (0.2 μm) absorbed by vegetable (i.e. wheat and lettuce) roots and transferred into shoots have induced negative effects on crop health via the alteration of the cell membrane and shifts in intracellular metabolism (Li et al., 2020). Nanoplastics could therefore enter the wider food chain by ending up in plant parts intended for human or livestock consumption (Bouwmeester et al., 2015). One study by Shi et al. (2019) also found that common plasticizers such as phthalates can be end up in wheat grains, exerting potential health risks to humans. Therefore, to preserve safe food production, the impacts of MPs especially nanoplastics, as well as the fate of MPs-derived chemical components on plant growth in agroecosystems deserve further attention.

4. Effects of MPs on soil C and nutrient cycling

4.1. Effect of MPs on belowground C inputs

MPs are mostly composed of C (e.g., polystyrene or polyethylene are almost 90% C), thus their incorporation into soil can represent a major source of non-plant derived C (Fig. 4; Rillig, 2018; Rillig and Lehmann, 2020). Regardless of their inherent properties, and if they are not lost through leaching or surface runoff, MPs gradually being immobilized, binding with soil minerals or organic compounds through biotic and abiotic processes. These C compounds could then become locked up within soil aggregates, physically protected from microbial decomposition (de Souza Machado et al., 2019), facilitating the formation of high molecular weight molecular (i.e. aromatic) compounds, subsequently altering SOM storage. Some authors have argued that MPs should be considered as part of SOM (Chen et al., 2020), which means that non-plastic SOM could be greatly overestimated under gross MPs pollution. MPs addition can also increase the amount of DOC in soils (Liu et al., 2017; Zang et al., 2020). Nanoplastics themselves contributed to between 9.78 and 21.21 mg L^{-1} DOC (Hu et al., 2019), which was in agreement with the changes in C storage in aquatic environments caused by MPs accumulation (Giering et al., 2014; Cole et al., 2016). Therefore, MPs-derived C could make a hidden contribution to soil C storage in SOM and DOM pools, especially considering the input of relative bioavailable C during the breakdown of biodegradable MPs in the microplastisphere.

MPs may indirectly alter plant below-ground C allocation. The altered N immobilization discussed above could decrease soil nutrient availability (Fig. 3), which increase belowground C inputs via photosynthesis (Zang et al., 2017). As the root constitutes a substantial proportion of SOM (Rasse et al., 2005; Zang et al., 2018), altered C allocation belowground can also occur due to the stimulated or suppressed plant growth caused by MPs pollution (de Souza Machado et al., 2019; Zang et al., 2019). For example, polyvinyl chloride (1–5%) powder increased wheat-derived C allocated into the soil due to a stimulation of root growth and enhanced rhizodeposition (Zang et al., 2020). However, we have also found that 10% polyethylene and polyvinyl chloride powders increased the C allocated to roots but decreased the amount of C incorporated into soil as indicated by ^{14}C phosphor imaging. As well as direct root exudation, C allocation to fungal symbionts (i.e. AMF) plays an another important role in C sequestration (Jones et al., 2009; Kaiser et al., 2015; Zhou et al., 2020a). Therefore, when AMF colonization rates changed with different types of MPs, this also alters rhizodeposition and the amount of photosynthetic C (Fig. 4). C input by plants into soil (i.e. rhizodeposition) is a major flux in the global C cycle compared to plant litter inputs, and is crucial not only for soil organic C sequestration, but also for microbial functions as a consequence for the maintenance of soil fertility and ecosystem stability. Therefore, a wider range of MPs types, sizes, shapes, and concentrations should be used in the future research to assess their effects on photosynthetic C allocation and subsequent net rhizodeposition in the plant-soil system.

4.2. Effect of MPs on SOM decomposition

Although 'conventional' petroleum-based MPs are largely non-bioavailable the material is not directly involved in SOM dynamics, however, they may regulate SOM decomposition through indirectly altering microbial processes. As discussed above, MPs can alter soil microbial community structures (Fig. 4; de Souza Machado et al., 2018b; Rillig et al., 2019a), affecting the turnover of native SOM. For example, the hydrophobic nature of polyester fiber can reduce water content and thus cause a better aerated environment around its surface (Guo et al., 2021), accelerating SOM mineralization (von Lützow et al., 2006). This is consistent with Liu et al. (2017) who demonstrated that 28% of polypropylene enhanced soil respiration by 3-folds, increased fluorescein diacetate hydrolase activity, and stimulated SOM decomposition. By contrast, the lower SOM decomposition under polyethylene fragments was associated with the degradation of soluble proteins and reduction of *Proteobacteria* abundance (de Souza Machado et al., 2018b; Wei et al., 2019). Given the lower persistence and easier degradation of biodegradable MPs, we speculate that enhanced bioavailable C resources from biodegradable MPs increases microbial activity, growth, and exoenzyme activity, potentially leading to the enhanced mineralization of native SOM by co-metabolism (i.e. microbial degradation of SOM using easily degradable polymers as an energy source), i.e. positive priming effect (Zhou et al., 2021). Therefore, we attribute the major effect of MPs on SOM decomposition to changes in soil properties and related microbial-mediated processes (Fig. 4). However, it should also be noted that MPs may also induce a negative priming effect due to the dilution and adsorption of soil available C (i.e. DOC) to their plastic surfaces according to organic-organic persistence hypothesis (Rillig et al., 2021). Further, it is likely that the effect may be shift to direct toxicity for the consequences of nanoplastics on SOM decomposition and C storage compared to MPs, as already shown for plants. Overall, the impact of MPs on native SOM decomposition is a new research topic, and the question of 'How will MPs with various sizes and shapes impact priming effect and C dynamics in agroecosystems?' still remains to be satisfactorily answered.

4.3. Effect of MPs on nutrient cycling and GHGs emission

Although MPs mostly contain negligible amounts of N and P, they can have significant effects on the microbial-mediated transformation of nutrients in soils. Soil nitrification was increased by MPs mainly due to increased soil porosity and oxygen diffusion (Green et al., 2016; de Souza Machado et al., 2018b; Chen et al., 2020). Moreover, antimicrobial properties of MPs may select for certain taxa (i.e. against nitrifiers, mycorrhizal symbiosis) and thus alter nutrients cycling (Beddow et al., 2017). For example, MPs altered AMF symbiosis and further impacted nutrient transport to plant roots (de Souza Machado et al., 2019). In addition, the suppression of ammonization, nitrification and denitrification processes was attributed to the absorption capacity of organic N due to the large specific surface area of MPs (Xia et al., 2016). It has been reported that polyacrylic acid could sequester NH_4^+-N by having carbonyl (=O) and hydroxyl (—OH) groups on the surface, directly reducing N availability (Chen et al., 2020). N cycling in soils may also be indirectly influenced by enzymes that hydrolyze SOM. The derivatives of polystyrene and polyethylene incorporated into soil can disrupt N cycling by limiting key N-related enzymes, e.g., chitinase and leucine aminopeptidase (Wang et al., 2016b; Bandopadhyay et al., 2019; Zang et al., 2020). Due to the positive effect on soil aggregation, polyester and polyethylene fibers (0.4%,) can increase the soil capacity to retain nutrients retention (i.e. N and P) (Lozano et al., 2021a). Overall, the direction and magnitude of nutrient dynamics under MPs remains unclear, and further experimentation is required.

Given that carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O) are the three most important climate-relevant greenhouse gases (GHGs) (Oertel et al., 2016), the evaluation of the effect of MPs on GHGs is of great importance. As discussed above, MPs induced changes in C and nutrients cycling in agroecosystems will result in variation in GHGs emissions. Added to this, because MPs may alter SOM mineralization (as shown in Section 4.2), one can expect MPs pollution to impact whether soils become a net CO_2 source or sink. The observed varying effects of MPs on N_2O and CH_4 production (Ren et al., 2020; Shen et al., 2020; Sun et al., 2020) can at least partially be explained by changes in the biophysical environment affecting the soil microbial population. MPs may increase water contents or decrease the porosity (de Souza Machado et al., 2018b; Boots et al., 2019), which could conceivably increase or decrease O_2 availability, resulting in incomplete denitrification processes thus modifying N_2O emissions (Jiang et al., 2016). Evidence for reduced N_2O emissions from agricultural soils was partially caused by lower N availability after polyethylene additions (Ren et al., 2020). The leaching of chemicals from MPs has also been shown to potentially contribute to the production of GHGs, such as CH_4 and ethylene (Romera-Castillo et al., 2018; Sun et al., 2020). However, polyvinyl chloride can decrease CH_4 emissions due to the suppression of hydrolysis-acidification and methanation (Wei et al., 2019). Many conclusions are still in the speculative stage, as there is insufficient data to support them. Given that GHGs are critical for global warming potential and future climate change, the potential effects of different MPs on GHGs emissions should become an integral part of future impact assessments.

5. Conclusions and future prospects

Microplastics are becoming widespread in many agroecosystems and their abundance is likely to increase for the foreseeable future, due to their continued input, inert properties, and slow degradation rates. It is unavoidable that MPs accumulation will impact plant and soil health, either by direct toxicity from additives and/or adsorbed contaminants or the potential to alter physico-chemical characteristics of the soils. However, the direction and magnitude of the impact are diverse and are dependent on the size, shape, type, and concentration of MPs. Although the C chains of MPs themselves are relatively inert, they can contribute to soil C storage, especially for easily biodegradable MPs with lower persistence. Furthermore, MPs may alter plant C allocation, and thus shift microbial communities and plant mycorrhizal symbiosis as well as causing an alteration in C, N and P-related enzymes, as a consequence is likely to affect the cycling of key nutrients and GHGs emissions. Surprisingly, biodegradable MPs appear to be a more potent inhibitor of plant growth and development, and an activator of SOM decomposition induced by priming effect due to the bioavailable C resources released with its degradation. To achieve a more accurate assessment of the effects of MPs on plant-soil health in agroecosystems, research priorities and directions for future research are proposed:

(1) The effect of MPs on soil properties requires further investigations considering the broad range of MPs found in soil. Along with MPs type, the effects of size, shape and concentration should also be considered. The mechanistic understanding of how MPs change soil properties needs to be addressed, since any changes in soil properties could impact the health of plants and wider microbial community. Additionally, the interactive

effects of MPs on soil aggregation and microbial communities should be taken into consideration.

(2) The formation and stability of SOM is vital for maintaining agroecosystem health and plays a major role in soil C and nutrient cycling (Lehmann and Kleber, 2015). As discussed, plastics come in a wide variety of forms containing a range of chemical components and additives (Rillig et al., 2019b), which may have various consequences for the health and sustainability of agroecosystems. Thus, there is an urgent need to thoroughly assess the likely effects of MPs as a function of their characteristics on rhizodeposition and SOM decomposition, as well as its underlying mechanisms. Furthermore, we should determine their contribution to soil C, by developing methods to quantify MPs, and MPs-derived C in soils globally. Critically, future studies must determine whether MPs adversely influence keystone microbial species (e.g., nitrifiers, AMF) that are fundamental to the major soil functions (e.g., SOM decomposition, nutrient cycling, litter decomposition, GHGs emissions).

(3) Although biodegradable MPs have been heralded as a solution to petroleum-based plastics, our review indicates that it is important to consider the potential disbenefits of such bioplastics, e.g., for plant growth and health. This is exemplified in the applications of plastic microbeads in cosmetics and plastic mulch films in agriculture where the negative environmental consequences have only been realized decades after their introduction (Rochman, 2018; Qi et al., 2020). As with other materials added to the soil (e.g., biochar), the effects on soil functions may be multifactorial related to changes in the physical, chemical, and biological soil properties. In-field testing of the degradation of biodegradable MPs under different scenarios (e.g., soil types, agricultural practice, and climate change) as well as using a realistic mixture of polymers over longer periods is therefore required, with particular attention to the effects on plant-microbe-soil interactions.

(4) Currently, the global and regional data inventory for the MPs pollution in agroecosystems is rare, and more detailed investigations are required. In future studies, researchers should extend the qualitative and quantitative evaluation of MPs in agroecosystems with various cropping systems under different agricultural practices, especially in the rhizosphere with microbial hotspots. Furthermore, global change is inherently a multifactorial phenomenon, especially on agroecosystems where multiple drivers co-occur with MPs contamination, such as N deposition, drought, and climate warming. Therefore, it is urgent to know how MPs interact with other evolutionary drivers in the agroecosystems affecting soil microbial functions and soil-plant health.

Declaration of competing interest

The authors declare that they have no known competing financial interests or personal relationships that could have appeared to influence the work reported in this paper.

Acknowledgments

This study was financially supported by the National Natural Science Foundation of China (31901470), the Fundamental Research Funds for the Central Universities (2020TC120) and a UK Global Challenges Research Fund project (NE/V005871/1) supported under the Natural Environment Research Council's Reducing the Impacts of Plastic Waste in Developing Countries Programme.

References

Alimi, O.S., Farner Budarz, J., Hernandez, L.M., Tufenkji, N., 2018. Microplastics and nanoplastics in aquatic environments: aggregation, deposition, and enhanced contaminant transport. Environ. Sci. Technol. 52, 1704–1724.

Andrady, A.L., 2011. Microplastics in the marine environment. Mar. Pollut. Bull. 62, 1596–1605.

Bandmann, V., Müller, J.D., Kohler, T., Homann, U., 2012. Uptake of fluorescent nano beads into BY2-cells involves clathrin-dependent and clathrin-independent endocytosis. FEBS Lett. 586, 3626–3632.

Bandopadhyay, S., Sintim, H.Y., DeBruyn, J.M., 2019. Structural and functional responses of soil microbial communities to biodegradable plastic film mulching in two agroecosystems. bioRxiv https://doi.org/10.1101/650317.

Beddow, J., Stolpe, B., Cole, P.A., Lead, J.R., Sapp, M., Lyons, B.P., Colbeck, I., Whitby, C., 2017. Nanosilver inhibits nitrification and reduces ammoniaoxidising bacterial but not archaeal amoA gene abundance in estuarine sediments. Environ. Microbiol. 19, 500–510.

Bolan, N., Kirkham, M.B., Halsband, C., Nugegoda, D., Ok, Y.S., 2020. Particulate Plastics in Terrestrial and Aquatic Environments. CRC Press ISBN 9780367511401.

Boots, B., Russell, C.W., Green, D.S., 2019. Effects of microplastics in soil ecosystems: above and below ground. Environ. Sci. Technol. 19, 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.

Bouwmeester, H., Hollman, P.C.H., Peters, R.J.B., 2015. Potential health impact of environmentally released micro- and nanoplastics in the human food production chain: experiences from nanotoxicology. Environ. Sci. Technol. 49, 8932–8947.

Brodhagen, M., Goldberger, J.R., Hayes, D.G., Inglis, D.A., Marsh, T.L., Miles, C., 2017. Policy considerations for limiting unintended residual plastic in agricultural soils. Environ. Sci. Pol. 69, 81–84.

Bünemann, E.K., Bongiorno, G., Bai, Z., Creamer, R.E., De Deyn, G., de Goede, R., Fleskens, L., Geissen, V., Kuyper, T.W., Mäder, P., Pulleman, M., Sukkel, W., van Groenigen, J.W., Brussaard, L., 2018. Soil quality—a critical review. Soil Biol. Biochem. 120, 105–125.

Cao, D., Wang, X., Luo, X., Liu, G., Zheng, H., 2017. Effects of polystyrene microplastics on the fitness of earthworms in an agricultural soil. IOP Conf. Ser. Earth Environ. Sci. 61, 12148.

Chen, H., Wang, Y., Sun, X., Peng, Y., Xiao, L., 2020. Mixing effect of polylactic acid microplastic and straw residue on soil property and ecological function. Chemosphere 243, 125271.

Cole, M., Lindeque, P.K., Fileman, E., Clark, J., Lewis, C., Halsband, C., Galloway, T.S., 2016. Microplastics alter the properties and sinking rates of zooplankton faecal pellets. Environ. Sci. Technol. 50, 3239–3246.

Du, F., Cai, H., Zhang, Q., Chen, Q., Shi, H., 2020. Microplastics in take-out food containers. J. Hazard. Mater. 399, 122969.

Fei, Y., Huang, S., Zhang, H., Tong, Y., Wen, D., Xia, X., Wang, H., Luo, Y., Barceló, 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.

Geyer, R., Jambeck, J.R., Law, K.L., 2017. Production, use, and fate of all plastics ever made. Sci. Adv. 19, e1700782.

Giering, S.L., Sanders, R., Lampitt, R.S., Anderson, T.R., Tamburini, C., Boutrif, M., Zubkov, M.V., Marsay, C.M., Henson, S.A., Saw, K., 2014. Reconciliation of the carbon budget in the ocean's twilight zone. Nature 507, 480–483.

Green, D.S., Boots, B., Sigwart, J., Jiang, S., Rocha, C., 2016. Effects of conventional and biodegradable microplastics on a marine ecosystem engineer (*Arenicola marina*) and sediment nutrient cycling. Environ. Pollut. 208, 426–434.

Guo, Q.Q., Xiao, M.R., Ma, Y., Niu, H., Zhang, G.S., 2021. Polyester microfiber and natural organic matter impact microbial communities, carbon-degraded enzymes, and carbon accumulation in a clayey soil. J. Hazard. Mater. 405, 124701.

Hahladakis, J.N., Velis, C.A., Weber, R., Iacovidou, E., Purnell, P., 2018. An overview of chemical additives present in plastics: migration, release, fate and environmental impact during their use, disposal and recycling. J. Hazard. Mater. 344, 179–199.

He, L., Wu, D., Rong, H., Li, M., Tong, M., Kim, H., 2018. Influence of nano- and microplastic particles on the transport and deposition behaviors of bacteria in quartz sand. Environ. Sci. Technol. 52, 11545–11554.

Heuchan, S.M., Fan, B., Kowalski, J.J., Gillies, E.R., Henry, H.A.L., 2019. Development of fertilizer coatings from polyglyoxylatepolyester blends responsive to root-driven pH change. J. Agric. Food Chem. 67, 12720–12729.

Hodson, M.E., Duffus-Hodson, C.A., Clark, A., Prendergast-Miller, M.T., Thorpe, K.L., 2017. Plastic bag derived-microplastics as a vector for metal exposure in terrestrial invertebrates. Environ. Sci. Technol. 51, 4714–4721.

Hu, D., Shen, M., Zhang, Y., Zeng, G., 2019. Micro(nano)plastics: an un-ignorable carbon source? Sci. Total Environ. 657, 108–110.

Huang, Y., Zhao, Y., Wang, J., Zhang, M., Jia, W., Qin, X., 2019. LDPE microplastic films alter microbial community composition and enzymatic activities in soil. Environ. Pollut. 254, 112983.

Huerta Lwanga, E., Gertsen, H., Gooren, H., Peters, P., Salánki, T., Van, D.P.M., Besseling, E., Koelmans, A.A., Geissen, V., 2016. Microplastics in the terrestrial ecosystem: implications for *Lumbricus terrestris* (Oligochaeta, Lumbricidae). Environ. Sci. Technol. 50, 2685–2691.

Huerta Lwanga, E., Gertsen, H., Gooren, H., Peters, P., Salanki, T., van der Ploeg, M., Besseling, E., Koelmans, A.A., Geissen, V., 2017. Incorporation of microplastics from litter into burrows of *Lumbricus terrestris*. Environ. Pollut. 220, 523–531.

Jambeck, J.R., Geyer, R., Wilcox, C., Siegler, T.R., Perryman, M., Andrady, A., 2015. Marine pollution. Plastic waste inputs from land into the ocean. Science 347, 768–771.

Jassby, D., Su, Y., Kim, C., Ashworth, V., Adeleye, A.S., Rolshausen, P., Roper, C., White, J., 2019. Delivery, uptake, fate, and transport of engineered nanoparticles in plants: a critical review and data analysis. Environ. Sci. Nano. 6, 2311–2331.

Jeong, C.B., Won, E.J., Kang, H.M., Lee, M.C., Hwang, D., Hwang, U.K., Lee, J.S., 2016. Microplastic size-dependent toxicity, oxidative stress induction, and p-JNK and p-p38 activation in the monogonont rotifer (*Brachionus koreanus*). Environ. Sci. Technol. 50, 8849–8857.

Jiang, T., Ma, X., Tang, Q., Yang, J., Li, G., Schuchardt, F., 2016. Combined use of nitrification inhibitor and struvite crystallization to reduce the NH_3 and N_2O emissions during composting. Bioresour. Technol. 217, 210–218.
Jiang, X., Chen, H., Liao, Y., Ye, Z., Li, M., Klobučar, G., 2019. Ecotoxicity and genotoxicity of polystyrene microplastics on higher plant *Vicia faba*. Environ. Pollut. 250, 831–838.
Jones, D.L., Nguyen, C., Finlay, R.D., 2009. Carbon flow in the rhizosphere: carbon trading at the soil–root interface. Plant Soil 321, 5–33.
Jones, D.L., Cooledge, E.C., Hoyle, F.C., Griffiths, R.I., Murphy, D.V., 2019. pH and exchangeable aluminum are major regulators of microbial energy flow and carbon use efficiency in soil microbial communities. Soil Biol. Biochem. 138, 107584.
Judy, J.D., Williams, M., Gregg, A., Oliver, D., Kumar, A., Kookana, R., 2019. Microplastics in municipal mixed-waste organic outputs induce minimal short to long-term toxicity in key terrestrial biota. Environ. Pollut. 252, 522–531.
Kaiser, C., Kilburn, M.R., Clode, P.L., Fuchslueger, L., Koranda, M., Cliff, J.B., Solaiman, Z.M., Murphy, D.V., 2015. Exploring the transfer of recent plant photosynthates to soil microbes: mycorrhizal pathway vs direct root exudation. New Phytol. 205, 1537–1551.
Kuzyakov, Y., Xu, X.L., 2013. Competition between roots and microorganisms for nitrogen: mechanisms and ecological relevance. New Phytol. 198, 656–669.
Lehmann, A., Zheng, W., Rillig, M.C., 2017. Soil biota contributions to soil aggregation. Nat. Ecol. Evol. 1, 1828–1835.
Lehmann, J., Kleber, M., 2015. The contentious nature of soil organic matter. Nature 528, 60–68.
Lei, L., Liu, M., Song, Y., Lu, S., Hu, J., Cao, C., Xie, B., Shi, H., He, D., 2018. Polystyrene (nano) microplastics cause size-dependent neurotoxicity, oxidative damage and other adverse effects in Caenorhabditis elegans. Environ. Sci. Nano. 5, 2009–2020.
Li, J., Zhang, K., Zhang, H., 2018. Adsorption of antibiotics on microplastics. Environ. Pollut. 237, 460–467.
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., 2020. Effective uptake of submicrometre plastics by crop plants via a crack-entry mode. Nat. Sustain. 3, 929–937.
Liu, E., He, W.Q., Yan, C., 2014. 'White revolution' to 'white pollution' — agricultural plastic film mulch in China. Environ. Res. Lett. 9, 091001.
Liu, H.F., Yang, X.M., Liu, G.B., Liang, C.T., 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, 907–917.
Lozano, Y.M., Rillig, M.C., 2020. Effects of microplastic fibers and drought on plant communities. Environ. Sci. Technol. 54, 6166–6173.
Lozano, Y.M., Aguilar-Trigueros, C.A., Onandia, G., Maaß, S., Zhao, T., Rillig, M.C., 2021a. Effects of microplastics and drought on soil ecosystem functions and multifunctionality. J. Appl. Ecol. 00, 1–9.
Lozano, Y.M., Lehnert, T., Linck, L.T., Lehmann, A., Rillig, M.C., 2021b. Microplastic shape, polymer type, and concentration affect soil properties and plant biomass. Front. Plant Sci. 12, 169.
von Lützow, M., Kogel-Knabner, I., Ekschmitt, K., Matzner, E., Guggenberger, G., Marschner, B., Flessa, H., 2006. Stabilization of organic matter in temperate soils: mechanisms and their relevance under different soil conditions—a review. Eur. J. Soil Sci. 57, 426–445.
Miyazaki, J., Kuriyama, Y., Tokumoto, H., Konishi, Y., Nomura, T., 2015. Cytotoxicity and behavior of polystyrene latex nanoparticles to budding yeast. Colloids Surf. A Physicochem. Eng. Asp. 469, 287–293.
National Bureau of statistics-China (NBSC), 2018. http://www.stats.gov.cn/.
Nizzetto, L., Bussi, G., Futter, M.N., Butterfield, D., Whitehead, P.G., 2016. A theoretical assessment of microplastic transport in river catchments and their retention by soils and river sediments. Environ. Sci. Process Impact. 18, 1050–1059.
Nomura, T., Tani, S., Yamamoto, M., Nakagawa, T., Toyoda, S., Fujisawa, E., Yasui, A., Konishi, Y., 2016. Cytotoxicity and colloidal behavior of polystyrene latex nanoparticles toward filamentous fungi in isotonic solutions. Chemosphere 149, 84–90.
Oertel, C., Matschullat, J., Zurba, K., Zimmermann, F., Erasmi, S., 2016. Greenhouse gas emissions from soils—a review. Chem. Erde-Geochem. 76, 327–352.
Plastics Europe, 2019. Plastics—the facts. February 4. Retrieved from plasticseurope.org. https://www.plasticseurope.org/application/files/5515/1689/9220/2014plastics_the_facts_PubFeb2015.pdf.
Prata, J.C., da Costa, J.P., Lopes, I., Duarte, A.C., Rocha-Santos, T., 2020. Environmental exposure to microplastics: an overview on possible human health effects. Sci. Total Environ. 702, 134455.
Prendergast-Miller, M.T., et al., 2019. Polyester-derived microfibre impacts on the soil-dwelling earthworm Lumbricus terrestris. Environ. Pollut. 251, 453–459.
Qi, Y., Ossowicki, A., Yang, X., Lwanga, E.H., Dini-Andreote, F., Geissen, V., Garbeva, P., 2020. Effects of plastic mulch film residues on wheat rhizosphere and soil properties. J. Hazard. Mater. 387, 121711.
Qi, Y.L., Yang, X.M., Pelaez, A.M., Lwanga, E.H., Beriot, N., Gertsen, H., Garbeva, P., Geissen, V., 2018. Macro- and micro-plastics in soil-plant system: effects of plastic mulch film residues on wheat (*Triticum aestivum*) growth. Sci. Total Environ. 645, 1048–1056.
Rasse, D.P., Rumpel, C., Dignac, M.F., 2005. Is soil carbon mostly root carbon? Mechanisms for a specific stabilisation. Plant Soil 269, 341–356.
Ren, X., Tang, J., Liu, X., Liu, Q., 2020. Effects of microplastics on greenhouse gas emissions and the microbial community in fertilized soil. Environ. Pollut. 256, 113347.
Rillig, M.C., 2018. Microplastic disguising as soil carbon storage. Environ. Sci. Technol. 52, 6079–6080.
Rillig, M.C., Lehmann, A., 2020. Microplastic in terrestrial ecosystems. Science 368, 1430–1431.
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., Ziersch, L., Hempel, S., 2017b. Microplastic transport in soil by earthworms. Sci. Rep. 7, 1362.
Rillig, M.C., Lehmann, A., de Souza Machado, A.A., Yang, G., 2019a. Microplastic effects on plants. New Phytol. 223, 1066–1070.
Rillig, M.C., Lehmann, A., Ryo, M., Bergmann, J., 2019b. Shaping up: toward considering the shape and form of pollutants. Environ. Sci. Technol. 53, 7925–7926.
Rillig, M.C., Leifheit, E., Lehmann, J., 2021. Microplastic effects on carbon cycling processes in soils. PLoS Biol. 19, e3001130.
Rochman, C.M., 2018. Microplastics research — from sink to source. Science 360, 28–29.
Romera-Castillo, C., Pinto, M., Langer, T.M., Álvarez-Salgado, X.A., Herndl, G.J., 2018. Dissolved organic carbon leaching from plastics stimulates microbial activity in the ocean. Nat. Commun. 9, 1–7.
Rousk, J., Brookes, P.C., Bååth, E., 2009. Contrasting soil pH effects on fungal and bacterial growth suggest functional redundancy in carbon mineralization. Appl. Environ. Microbiol. 75, 1589–1596.
Sander, M., 2019. Biodegradation of polymeric mulch films in agricultural soils: concepts, knowledge gaps, and future research directions. Environ. Sci. Technol. 53, 2304–2315.
Seeley, M.E., Song, B., Passie, R., Hale, R.C., 2020. Microplastics affect sedimentary microbial communities and nitrogen cycling. Nat. Commun. 11, 2372.
Shen, M., Zhu, Y., Zhang, Y., Zeng, G., Wen, X., Yi, H., Ye, S., Ren, X., Song, B., 2019. Micro (nano)plastics: unignorable vectors for organisms. Mar. Pollut. Bull. 139, 328–331.
Shen, M., Huang, W., Chen, M., Song, B., Zeng, G., Zhang, Y., 2020. (Micro)plastic crisis: unignorable contribution to global greenhouse gas emissions and climate change. J. Clean. Prod. 254, 120138.
Shi, M., Sun, Y., Wang, Z., He, G., Quan, H., He, H., 2019. Plastic film mulching increased the accumulation and human health risks of phthalate esters in wheat grains. Environ. Pollut. 250, 1–7.
de Souza Machado, A.A., Kloa, s 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., Till, J., Kloas, K., Lehmann, A., Becker, R., Rillig, M.C., 2018b. Impacts of microplastics on the soil biophysical environment. Environ. Sci. Technol. 52, 9656–9665.
de Souza Machado, A.A., Lau, C.W., Kloas, W., Bergmann, J., Bachelier, J.B., Faltin, E., Becker, R., Görlich, 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., Horton, A.A., Davis, T., Maaß, S., 2020. Microplastics and their effects on soil function as a life-supporting system. In: He, D., Luo, Y. (Eds.), Microplastics in Terrestrial Environments. The Handbook of Environmental Chemistry. vol. 95. Springer, Cham.
Sintim, H.Y., Flury, M., 2017. Is biodegradable plastic mulch the solution to agriculture'splastic problem? Environ. Sci. Technol. 51, 1068–1069.
Sun, Y., Ren, X., Pan, J., Zhang, Z., Tsui, T.H., Luo, L., Wang, Q., 2020. Effect of microplastics on greenhouse gas and ammonia emissions during aerobic composting. Sci. Total Environ. 737, 139856.
Teuten, E.L., Saquing, J.M., Knappe, D.R.U., Barlaz, M.A., et al., 2009. Transport and release of chemicals from plastics to the environment and to wildlife. Philos. Trans. R. Soc. B Biol. Sci. 364, 2027–2045.
Volova, T.G., Prudnikova, S.V., Vinogradova, O.N., Syrvacheva, D.A., Shishatskaya, E.I., 2017. Microbial degradation of polyhydroxyalkanoates with different chemical compositions and their biodegradability. Microb. Ecol. 73, 353–367.
Wan, Y., Wu, C., Xue, Q., Hui, X., 2019. Effects of plastic contamination on water evaporation and desiccation cracking in soil. Sci. Total Environ. 654, 576–582.
Wang, J., Tan, Z., Peng, J., Qiu, Q., Li, M., 2016a. The behaviors of microplastics in the marine environment. Mar. Environ. Res. 113, 7–17.
Wang, J., Lv, S., Zhang, M., Chen, G., Zhu, T., Zhang, S., 2016b. Effects of plastic film residues on occurrence of phthalates and microbial activity in soils. Chemosphere 151, 171–177.
Wang, W.F., Ge, J., Yu, X.Y., Li, H., 2020. Environmental fate and impacts of microplastics in soil ecosystems: progress and perspective. Sci. Total Environ. 708, 134841.
Wei, W., Huang, Q.S., Sun, J., Wang, J.Y., Wu, S.L., Ni, B.J., 2019. Polyvinyl chloride microplastics affect methane production from the anaerobic digestion of waste activated sludge through leaching toxic bisphenol-A. Environ. Sci. Technol. 53, 2509–2517.
Weithmann, N., Möller, J.N., Löder, M.G., Piehl, S., Laforsch, C., Freitag, R., 2018. Organic fertilizer as a vehicle for the entry of microplastic into the environment. Sci. Adv. 4, e8060.
Xia, X., Liu, T., Yang, Z., Michalski, G., Liu, S., Jia, Z., Zhang, S., 2016. Enhanced nitrogen loss from rivers through coupled nitrification-denitrification caused by suspended sediment. Sci. Total Environ. 579, 47–59.
Yu, Q., Hu, X., Yang, B., Zhang, G., Wang, J., Ling, W., 2020. Distribution, abundance and risks of microplastics in the environment. Chemosphere 249, 126059.
Zang, H., Blagodatskaya, E., Wen, Y., Xu, X., Kuzyakov, Y., 2018. Carbon sequestration and turnover in soil under the energy crop *Miscanthus*: repeated ^{13}C natural abundance approach and literature synthesis. GCB Bioenergy 10, 262–271.
Zang, H., Xiao, M., Wang, Y., Ling, N., Wu, J., Ge, T., Kuzyakov, Y., 2019. Allocation of assimilated carbon in paddies depending on rice age, chase period and N fertilization: experiment with $^{13}CO_2$ labelling and literature synthesis. Plant Soil 445, 113–123.
Zang, H., Zhou, J., Marshall, M.R., Chadwick, D.R., Wen, Y., Jones, D.L., 2020. Microplastics in the agroecosystem: are they an emerging threat to the plant-soil system? Soil Biol. Biochem. 148, 107926. https://doi.org/10.1016/j.soilbio.2020.107926.
Zang, H.D., Blagodatskaya, E., Wang, J.Y., Xu, X.L., Kuzyakov, Y., 2017. Nitrogen fertilization increases rhizodeposit incorporation into microbial biomass and reduces soil organic matter losses. Biol. Fertil. Soils 53, 419–429.
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., 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, S.L., Wang, J.Q., Yan, P.K., Hao, X.H., Xu, B., Wang, W., Aurangzeib, M., 2020. Non-biodegradable microplastics in soils: a brief review and challenge. J. Hazard. Mater. 409, 124525. https://doi.org/10.1016/j.jhazmat.2020.124525.
Zhou, J., Zang, H.D., Loeppmann, S., Gube, M., Kuzyakov, Y., Pausch, J., 2020a. Arbuscular mycorrhiza enhances rhizodeposition and reduces the rhizosphere priming effect on the decomposition of soil organic matter. Soil Biol. Biochem. 140, 107641.
Zhou, J., Wen, Y., Shi, L.L., Marshall, M.R., Kuzyakov, Y., Blagodatskaya, E., Zang, H.D., 2020b. Strong priming of soil organic matter induced by frequent input of labile carbon. Soil Biol. Biochem. 152, 108069.
Zhou, J., Gui, H., Banfield, C.C., Wen, Y., Zang, H.D., Dippold, M.A., Charlton, A., Jones, D.L., 2021. The microplastisphere: biodegradable microplastics addition alters soil microbial community structure and function. Soil Biol. Biochem. 156, 108211. https://doi.org/10.1016/j.soilbio.2021.108211.
Zhu, D., Chen, Q.L., An, X.L., Yang, X.R., Christie, P., Ke, X., Zhu, Y.G., 2018. Exposure of soil collembolans to microplastics perturbs their gut microbiota and alters their isotopic composition. Soil Biol. Biochem. 116, 302–310.
Zubris, K.A.V., Richards, B.K., 2005. Synthetic fibers as an indicator of land application of sludge. Environ. Pollut. 138, 201–211.