Chen et al. Carbon Research (2024) 3:37
https://doi.org/10.1007/s44246-024-00124-1

Carbon Research

REVIEW

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Effects of microplastics on soil carbon pool and terrestrial plant performance

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Abstract

Soil, as a primary repository of plastic debris, faces an escalating influx of microplastics. Microplastics have the potential to decrease soil bulk density and pH, as well as alter soil pore structure and aggregation. These changes in soil physico-chemical properties subsequently lead to habitat degradation for microbes and environmental shifts that impact plant growth. Masquerading as soil carbon storage, microplastics can distort assessments of the soil carbon pool by introducing plastic-carbon and associated leachates, influencing soil organic matter (SOM) turnover through priming effects (e.g., dilution, substrate switching, and co-metabolisms). Additionally, microplastics can influence the distribution of soil carbon in particulate and mineral-associated organic matter, consequently affecting the accumulation and stability of soil carbon. Furthermore, microplastics can also influence the chemodiversity of dissolved organic matter (DOM) in soils by increasing DOM aromaticity and molecular weight while deepening its humification degree. The changes observed in soil DOM may be attributed to inputs from microplastic-derived DOM along with organo-organic and organo-mineral interactions coupled with microbial degradation processes. Acting as an inert source of carbon, microplastics create a distinct ecological niche for microbial growth and contribute to necromass formation pathways. Conventional microplastics can reduce microbial necromass carbon contribution to the stable pool of soil carbon, whereas bio-microplastics tend to increase it. Furthermore, microplastics exert a wide range of effects on plant performance through both internal and external factors, influencing seed germination, vegetative and reproductive growth, as well as inducing ecotoxicity and genotoxicity. These impacts may arise from alterations in the growth environment or the uptake of microplastics by plants. Future research should aim to elucidate the impact of microplastics on microbial necromass accumulation and carbon storage within mineral-associated fractions, while also paying closer attention to rhizosphere dynamics such as the microbial stabilization and mineral protection for rhizodeposits within soils.

Highlights

- Microplastics (MPs) have either positive or negative effects on SOM mineralization.
- MPs affect soil carbon distribution in particulate and mineral-associated fraction.
- MPs increase the aromaticity, molecular weight and humification degree of soil DOM.
- Conventional MPs can reduce microbial necromass, whereas bio-MPs cannot.
- MPs influence plant performance through both internal and external factors.

Keywords Microplastic, Soil organic matter, Priming effects, Mineral associated organic matter, Dissolved organic matter, Microbial necromass carbon, Rhizosphere

Handling editor: Fangbai Li.

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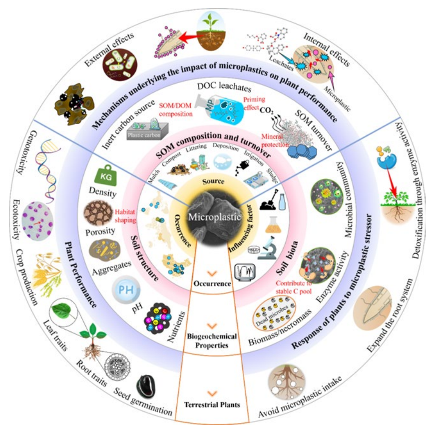
Graphical Abstract



1 Introduction

The terrestrial environment serves as a significant reservoir for microplastics, receiving 4–23 times more plastic waste annually compared to the marine environment (de Souza Machado et al. 2018a; Nizzetto et al. 2016). However, due to the challenges associated with separating microplastics from soil organic matter (SOM) and minerals, it was not until Rillig's call for research in 2012 that scientific attention began to focus on surveying microplastics in soil (Rillig 2012). Subsequently, the scientific community discovered the presence of microplastics in various soils worldwide, including home gardens, greenhouses, agricultural lands, coastal areas, industrial sites, and floodplain soils (Table S1). Based on our global

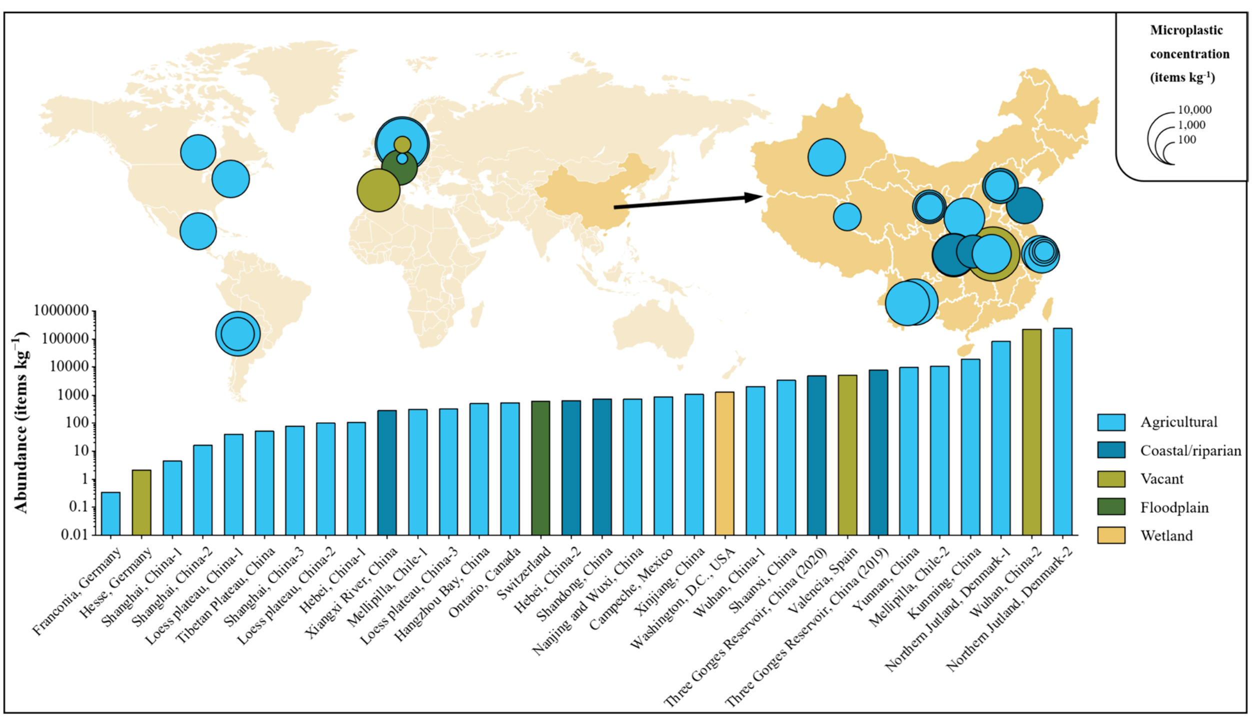


Fig. 1 Global microplastics abundance in different soils. Here, the data expressed as numerical concentration (items per kilogram soil) were ranked and mapped, whereas global comparison of data expressed as mass concentration (mg per kilogram soil) were provided in Fig. S1. Details information about the sampling and processing methods were provided in Table S1

inventory of microplastics in soil environments, recovered microplastic concentrations range widely from baseline levels up to 20 mg kg^{-1} in inhabited areas and reach as high as 67,500 mg kg^{-1} in industrial soil (Figs. 1 and S1), while some studies have suggested even more severe levels of microplastic contamination that may occur in specific soils (Huerta Lwanga et al. 2017). The sources of soil microplastics primarily encompass sewage sludge amendment, irrigation, composting, plastic mulching, dry and wet deposition (i.e., rain and snow) from the atmosphere, fragmentation and degradation processes, the mismanaged runoff by sewer systems from roads or littering activities, landfills, as well as plastics-processing plants (Table S1) (Chen et al. 2022c; Chen et al. 2020; Chen et al. 2023a; Feng et al. 2020; van den Berg et al. 2020). Microplastics exhibit remarkable resistance to environmental degradation (de Souza Machado et al. 2018a; He et al. 2023) and can accumulate persistently in the soil environment over an extended time period while exerting continuous ecosystem effects (Gao et al. 2023; Geyer et al. 2017; Li et al. 2024a; Pan et al. 2023). The pervasive presence, high abundance, and non-natural and persistent characteristics of microplastics in soil underscore their non-negligible role in soil ecology and emphasize the urgency to address their adverse impacts.

In recent years, there has been a gradual shift in research focus towards investigating the ecosystem impacts of microplastics in soil (Rillig and Lehmann 2020). Upon entering the soil environment, microplastics have the potential to modify physicochemical properties, reshape microbial habitats, and alter resident microbial communities and activities, consequently influencing soil function and nutrient cycling. Regarding the carbon cycle in soils, microplastics can directly influence the composition of soil carbon pools by introducing both recalcitrant and labile carbon fractions such as soluble additives. Additionally, microplastics may induce positive or negative priming effects on SOM mineralization thereby impacting soil carbon stability (Zhang et al. 2023a). Notably, soil microbes play dual roles in mediating the soil carbon cycle by promoting canbon release through catabolic activities while also preventing canbon release via canbon stabilization mechanisms (Schimel and Schaeffer 2012). Microplastics could potentially affect microbial functions associated with SOM decomposition. Furthermore, microplastic interactions with microbial death pathways may regulate the accumulation and stability of microbial necromass (Camenzind et al. 2023; Chen et al. 2023b). Other mechanisms of direct microplastic interaction with microbes such as electrochemistry-based "electron shuttling" and "microbial frustration" might also impact the soil carbon cycle dynamics (Rillig et al. 2021). However, the impact of microplastics on soil carbon pool composition and stability remains largely unexplored, and the underlying mechanisms have yet to be comprehensively examined.

Soil plants play a crucial role in soil carbon and soil health by fixing carbon through photosynthesis. The rhizosphere of plant is a hotspot of microbial activity and significantly influences the soil carbon cycle. Existing evidence suggests that microplastics impact the rhizosphere environment, including aggregates and microbial communities (Rong et al. 2023; Song et al. 2023), thereby affecting the input of rhizodeposits into soil carbon pools and the stability of soil organic carbon (SOC). Moreover, plastic debris could accumulate pesticides and other toxins present in the soil and leach out additives with potential carcinogenicity and mutagenicity (Ramos et al. 2015), which may affect crop quality, thus posing a threat to human health through the food chain (Fu and Du 2011; Ge et al. 2021; Khalid et al. 2020; Li et al. 2024b; Wang et al. 2022, 2015a). In the context of the near-permanent increase in microplastic contamination, a holistic understanding of ecosystem feedbacks of soil to microplastic contamination is needed.

In the present review, we conducted an exhaustive literature review to address the following two topics: (1) the effects of microplastics on the content, composition, and turnover of SOM, especially the more sensitive dissolved organic matter (DOM) and the more stable microbial necromass carbon, and (2) the effects of microplastics on plant performance and potential mechanisms, as well as plant responses to microplastic stressors. For each topic, knowledge gaps and future prospects are proposed.

2 Effects of microplastics on soil physicochemical properties

Ample evidence from the scientific literature suggests that microplastics can modify soil physicochemical properties in many ways, including soil density, porosity, aggregation, pH, and fertility (Fig. 2). These property changes will subsequently shape the habitat for microbial growth, regulate microbial activity, mediate organic carbon bioaccessibility and turnover rate, and affect plant performance. The effects of microplastics on soil properties depend on polymer type, size class, shape, dose, and soil type.

2.1 Soil bulk density

Soil bulk density is generally associated with soil physical quality and rootability, and high soil bulk density usually means strong penetration resistance and poor root growth (Dexter 2004). The density of microplastics ranges from 0.88 to 1.70 g cm^{-3} (Table S2), which is generally less dense than the predominant minerals in different soils (Lincmaierová et al. 2023). Therefore, the

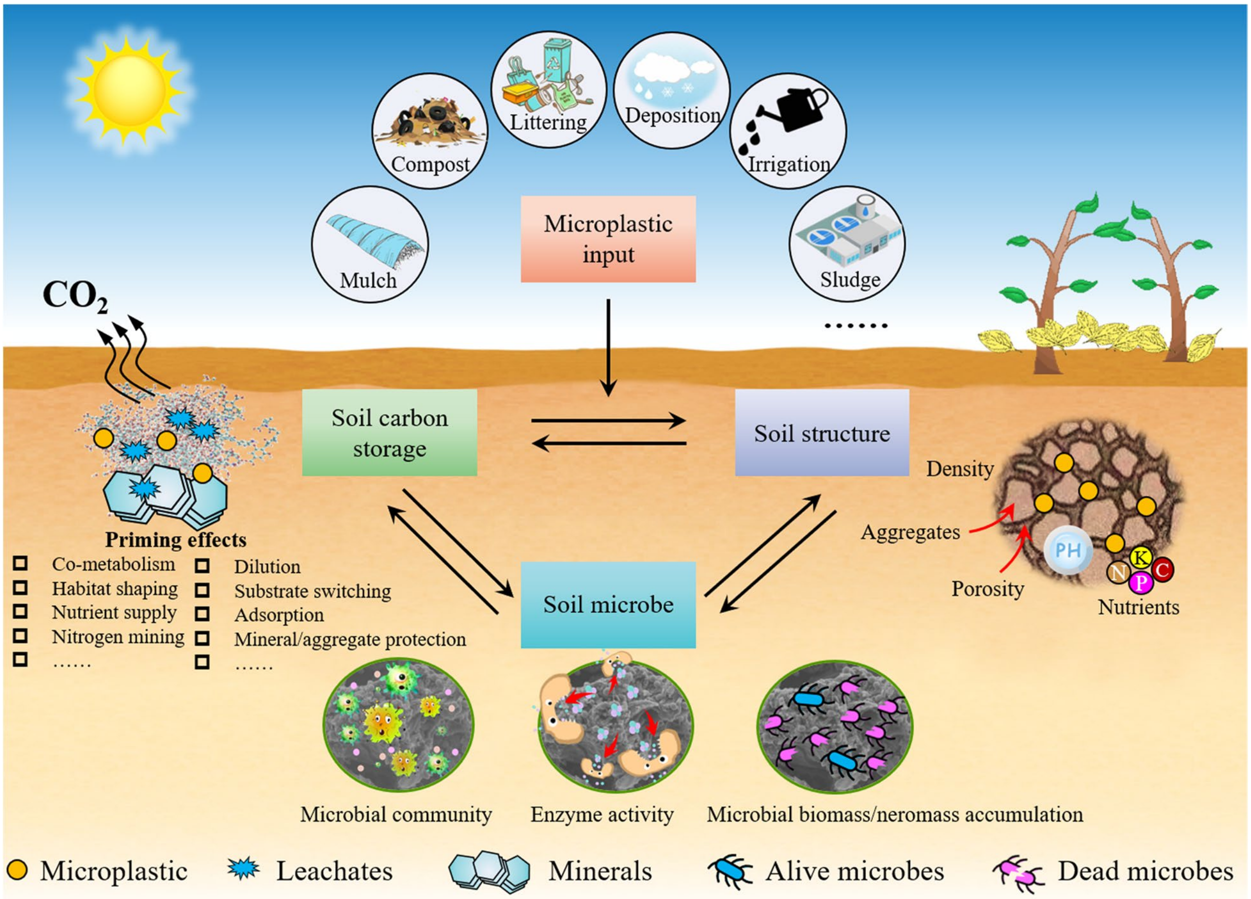


Fig. 2 The conceptual diagram depicting the impact of microplastics on soil structure, soil microbes, and turnover of SOM. Microplastics infiltrate the soil environment through various pathways, including mulching, composting, littering, atmospheric deposition, irrigation, and sludge application. These microplastics can alter physicochemical properties of the soil such as density, porosity, aggregation, pH levels, and nutrient availability. Consequently, these changes in properties shape the microbial habitat by influencing growth patterns and regulating activity while also mediating bioaccessibility and turnover rate of SOM. Soil microbes play a dual role in controlling SOM turnover; they promote carbon release through catabolic activities but also prevent its release via mechanisms like necromass production and stabilization. Microplastics may influence SOM mineralization through both positive and negative priming effects with mechanisms primarily involving co-metabolism, dilution effects, and substrate switching

involvement of microplastics often decreases the bulk density of soils. For example, de Souza Machado et al. (2018b) found that polyacrylic fibers, polyamide (PA) beads, polyester (PES) fibers, and polyethylene (PE) fragments all decreased the bulk density of loamy sand soils, with only PES fibers possessing a significant effect at an application rate of 0.4% (w/w) and having a concentration-dependent response. However, the decrease in soil bulk density by microplastic addition does not necessarily indicate positive effects on root growth, as it might not increase soil porosity. Moreover, microplastics could increase soil density in the rhizosphere despite decreasing bulk density, thus exerting complex effects on plant growth (de Souza Machado et al. 2019).

Although the majority of studies suggested a decrease in soil bulk density by microplastic addition, some also showed divergent results (Li et al. 2023). For example, Zhang et al. (2019) suggested that no detectable changes in soil bulk density were observed for clay loam soils at an application rate of 0.3% (w/w). Therefore, it is plausible to speculate that soil texture is a crucial factor regulating the influence of microplastics on soil properties. However, more evidence is needed to verify this speculation.

2.2 Soil porosity

Soil porosity plays a key role in regulating the infiltration rate of water into soil and the rate of nutrient loss from soil through water evaporation, as well as soil microbial respiration. In addition, small pores could reserve soil water and reduce leaching through capillary action (Major et al. 2009). Zhang et al. (2019) observed that PES fibers could alter the pore structure in clay loam soil. de Souza Machado et al. (2018b) demonstrated that PES fibers affected the soil water-holding capacity, which could further affect soil moisture and evapotranspiration (de Souza Machado et al. 2019). Wan et al. (2019) found that PE film could increase water evaporation from clay soil. Jiang et al. (2017) suggested that plastic-film residues (~15 kg ha^{-1}) could significantly alter the initial gravimetric water content, bulk density, total porosity, and soil water distribution. In addition, Wang et al. (2015c) demonstrated that due to their waterproofness, plastic-film residues could block soil pores and decrease soil pore size, thus affecting water infiltration and resulting in the loss of soil water. Moreover, changes in porosity could alter the sorption capacity of soils and affect the ability of microbes to reach and degrade potential organic substrates (Baldock and Skjemstad 2000). Such deviations from a natural state suggest the potential threat of microplastics to soil ecosystems (Guo et al. 2022).

2.3 Soil aggregates

Soil aggregates are the basis of soil function and play a crucial role in regulating soil structure, shaping the habitat for soil organisms, and determining SOM turnover (Bronick and Lal 2005; Tisdall and Oades 1982). In addition, soil aggregates also significantly affect soil porosity, which in turn influences the movement of gases and water and the activities of associated microbial communities (Rillig and Lehmann 2020; Rillig et al. 2017). The size and stability of soil aggregates regulate soil erodibility (López et al. 2000; Planchon et al. 2000; Somaratne and Smettem 1993). A decrease in soil aggregates might decrease the diversity of soil microenvironments, thereby impoverishing the soil structure (Six et al. 2006; Zheng et al. 2016). PA beads were more prone to be incorporated into the soil matrix (de Souza Machado et al. 2018b) and soil minerals (Chen et al. 2022d) than other polymer types, and PA beads are unlikely to be isolated from soils after incubations. In addition, fragment-shaped microplastics were loosely integrated into soil aggregates, while fiber-shaped microplastics were more tightly integrated (Guo et al. 2020). Thus, microplastics could integrate with soil to varying degrees, which further influences the formation of aggregates (Liu et al. 2023). Zhang and Liu (2018) reported that 72% of the recovered microplastics in soil were associated with soil aggregates and that fibrous microplastics were most commonly found in microaggregates. Boots et al. (2019) observed that the addition of high density polyethylene (HDPE), polylactic acid (PLA), and synthetic fiber all significantly decreased the mean size of water-stable soil aggregates irrespective of the presence of *Lolium perenne*. Similarly, the addition of polyacrylic fibers (0.05%, 0.10%, 0.20%, and 0.40%, w/w) and PES fibers (0.20% and 0.40%, w/w) all significantly lowered soil water-stable aggregates, and the effects were more significant at higher concentrations (de Souza Machado et al. 2018b). The decrease in soil water-stable aggregates thus increased the water-holding capacity and evapotranspiration. However, the effects of other types of microplastics, such as PA beads and PE fragments, on soil aggregates were much weaker, and low concentrations of PA and high concentrations of PE even exerted divergent results (de Souza Machado et al. 2018b). Moreover, Zhang et al. (2019) observed an increase in soil water-stable aggregates after the addition of 0.1% and 0.3% (w/w) PES fibers. The significant disparity among various studies can be attributed to the use of different polymer types and microplastic concentrations, as well as the influence of distinct soil textures and time duration (Chen et al. 2023b).

A meta-analysis demonstrated that soil microbes contributed to the formation of soil aggregation, although the positive correlation displayed considerable variability regarding the species investigated (Lehmann et al. 2017). A recent study showed that exposure to PE fragments increased the correlation between microbial activity and soil aggregation (de Souza Machado et al. 2018b). The shifts in the correlation between microbial activity and soil aggregation might suggest either a shift in the microbial community or an alteration in the decay of SOM (de Souza Machado et al. 2018b). However, further efforts are required to disentangle the possible underlying mechanisms among the changes in soil physiochemical properties, soil biota and SOM turnover.

2.4 Soil pH

Soil pH is one of the most important chemical attributes in soil and plays a central role in regulating soil properties and microbial succession (Aciego Pietri and Brookes 2008; Rousk et al. 2009; Wang et al. 2024). Moreover, it can also affect plant performance by directly regulating physiological activities in seed germination and root growth, as well as indirectly affecting microbial activity, nutrient availability, and precipitation and dissolution equilibrium through its effects on ion mobility (Bloom 2000). Bandow et al. (2017) reported a decrease in soil pH in the presence of HDPE pellets after 6 and 12 weeks of exposure to photo- and thermo-oxidative conditions. Boots et al. (2019) also observed that the pH of the soil was 0.62 units lower than that of the control after exposure to HDPE in soil planted with *Lolium perenne* for 30 days. Similarly, a decrease in pH (0.16–0.21 units) was observed in soil planted with *Oryza sativa L.* after the first supplementary fertilization (Chen et al. 2022b). Given the weak possibility of the photo-oxidation of microplastics in soil, it is plausible to speculate that the decreased pH was associated with the changed status of cation exchange in the soil and free exchange of protons in the soil water. Further work is required to verify this speculation.

2.5 Soil fertility and nutrient cycling

Microplastics can also affect the fertility and nutrient cycling of soil. A significant increase in the concentrations of DOM, inorganic nitrogen and total phosphorus was observed after 28% (w/w) microplastic treatment in a sandy loam soil, whereas no obvious changes were observed for 28% (w/w) microplastic treatment over 30 days (Liu et al. 2017). Similarly, bioplastic addition significantly increased the DOM and dissolved organic nitrogen content in soil, whereas no significant effects were observed for LDPE (Meng et al. 2022). Notably, the aforementioned treatments are far beyond the environmentally related microplastic concentrations, which may show divergent results from natural conditions. Thus, future studies should underline these phenomena regarding soils of different textures over a long time span. A field study on the contamination of plastic-mulching residues showed that SOM and alkali-hydrolysable N were greatly reduced after a 500 kg ha^{-1} microplastic application, and an increasing microplastic application rate of 2,000 kg ha^{-1} further decreased the available P (Hegan et al. 2015). These results confirmed the role of plastic debris in diminishing soil fertility (Xiang et al. 2023), although further work under environmentally related conditions should be conducted to better understand this impact. N-containing microplastics may serve as a nitrogen source and enrich the soil nitrogen content (Palmer 2001). This speculation can be partly supported by high N_2O emissions and N-acquiring enzyme activity (β-N-acetyl-glucosaminidase) in artificial soils using PA as the sole carbon source (Chen et al. 2022d).

3 Effects of microplastics on soil carbon pool

As microplastics are mainly composed of carbon, they may disguise as soil carbon storage and affect the content composition and turnover of SOM (Rillig and Lehmann 2020). However, the rate of microplastic carbon entering the soil and its ecosystem effects on SOM are still open questions.

3.1 Disguising as carbon storage and serving as an inert carbon source in soil

Most microplastics are composed of more than 90% carbon, among other elements (Rillig and Lehmann 2020). They are incorporated into soil aggregates in consort with minerals and organic matter. Microplastics can be disguised as part of soil carbon storage independent of photosynthesis and net primary production (Rillig and Bonkowski 2018). Due to its resistance to decomposition, microplastic carbon tends to accumulate in the soil and eventually be immobilized by microbes after a long time span, which may somehow contribute to the carbon cycle and affect SOM turnover (Chen et al. 2022d; Liu et al. 2017).

Many previous records have shown the microbial mineralization of microplastic carbon based on direct and indirect evidence, such as changes in surface roughness and functional groups, the diminishment of particle size, the loss of plastic weight, and the colonization of biofilms (McCormick et al. 2014; Yang et al. 2018). In the soil environment, the degradation of microplastics was proven to be very slow (Rillig and Bonkowski 2018). For example, only 0.1 to 0.4% weight loss was observed for PE after 800 days of burial in soil (Albertsson 1980). In addition, only 0.4% weight loss was observed for polypropylene (PP) after one year of incubation (Arkatkar et al. 2009). Moreover, no degradation was found for polyvinyl chloride (PVC) even after 10 to 35 years of burial in soil (Ali et al. 2014; Otake et al. 1995; Santana et al. 2012). However, Chen et al. (2022d) further monitored the CO_2 emissions derived from polymer carbon to investigate the accurate degradation pattern of microplastic carbon in artificial soil. They showed that the mineralization ratio of microplastic carbon ranged from 0.007% to 0.876% after 56 days, the upper limit of which was in close proximity to that of SOM (~ 1%) and even exceeded that of biochar-derived carbon (> 0.1%) (Yang et al. 2022). Despite the inert nature of polymer carbon, the results of this fundamental research indicated that the mineralization of polymer carbon is a non-negligible source of CO_2 emissions (Chen et al. 2022d) and should be taken into careful consideration in the assessment of carbon storage and carbon emissions (Rillig and Lehmann 2020). In particular, such effects are supposed to be more significant for emerging bioplastics with short lifespans (Zhang et al. 2023a).

In the real soil environment, however, it is difficult to distinguish whether CO_2 emissions are derived from microplastic carbon or native SOM. One possible solution is to use carbon isotope labeling to track the carbon source. Zumstein Michael et al. (2018) tracked the carbon from biodegradable polymers into CO_2 and microbial biomass in soil based on ^{13}C-labeled polymers and isotope-specific analytical methods. The polymer used in this study was a self-synthesized poly(butylene adipate-co-terephthalate) using ^{13}C-labeled monomers. Zhang et al. (2023a) distinguished the carbon source of CO_2 based on the distinct $\delta^{13}C$ values of poly-hydroxyalkanoates

(PHA), polybutylene succinate (PBS), and PLA from the soils. The observed priming effects across farmland soils were 552% − 1744% for PHA, 44% − 179% for PBS, and − 29% − 43% for PLA. The study underscores the influence of microplastic degradability on the amplitude of priming effects, highlighting the potential significance of microbial nitrogen mining in driving long-term priming effects, given the nitrogen deficiency associated with microplastics. For other commonly used plastic products, such as PE, the ^{13}C isotope labeling of the monomers seems not easy to accomplish. Thus far, it remains largely unknown how much microplastics contribute to carbon emissions.

3.2 Input of DOM leachates in the soil active carbon pool

Various additives, such as plasticizers, stabilizers, antioxidants, flame retardants, and pigments, are usually used to improve plastic quality. The most commonly used additives detected in the environment include phthalates, nonylphenols, bisphenol-A, and brominated flame retardants (Hermabessiere et al. 2017), many of which have been classified as endocrine disruptors and carcinogens (Cherif Lahimer et al. 2017, Wang et al. 2015a). In addition to plastic additives, DOM leachates also include the weathering products of plastics, which mainly comprise long-chain alkanes, high-molecular-weight acids, and benzoic ether (Zhang et al. 2022), which are associated with monomers and oligomers that initially form the polymer chain (Lee and Hur 2020).

These microplastic leachates can be released from microplastics and enter the soil system. On the one hand, leachates serve as an active carbon pool and are directly involved in the soil carbon cycle (Cristina et al. 2022). Romera-Castillo et al. (2018) estimated that global plastic leachates amounted to 23,600 metric tons in the marine environment annually, approximately 60% of which are accessible to microbial utilization in less than 5 days. A 56-day incubation suggested that microplastic leachates had higher degradability than riverine humic acid (Zhang et al. 2022). Thus, it is plausible to speculate that microplastic leachates may have a vital contribution to soil carbon emissions. On the other hand, leachates can be integrated into minerals and SOM (Lee and Hur 2020), exert extreme toxicity to soil microbes (Gaylor et al. 2013; Wei et al. 2019b), and alter the activity, diversity, community structure and metabolism of soil microbes, possibly by destroying the fluidity of the cell membrane (Xie et al. 2010; Zhou et al. 2005). The selection effects on soil biota suggested shifts in the SOM utilization pathway by microbes. Alternatively, leachates are somehow a more active carbon source than plastic solids and certain organic matter and may affect SOM turnover through priming effects (Chen et al. 2022d; Zhang et al. 2022). The priming effects of leached DOM from plastics may, in turn, instigate alterations in DOM across diverse soil types (Sun et al. 2022).

3.3 Changes in SOM composition and mineralization

As mentioned above, microplastics and their leachates could contribute to the inert and active soil carbon pool. Microplastics could be detected as carbon through the current methods applied to quantify SOM. Correspondingly, the SOM contents in soil are prone to increase after microplastic addition (Kim et al. 2021; Li et al. 2022a). However, some studies have also showed that the SOM content was not measurably affected (Boots et al. 2019; Meng et al. 2022) or even decreased (Meng et al. 2022) after microplastic treatments, which was potentially attributed to the low microplastic application rate and the mineralization of polymer carbon or SOC.

Several recent studies have noted shifts in the composition and turnover rate of SOM after microplastic application. For example, PES and bioplastics showed adverse effects on the abiotic characteristics of SOM, such as physical stability (indicated by water molecule bridges), water binding (indicated by decreased desorption enthalpy or faster desorption), and the stability of SOM aliphatic crystallites (Fojt et al. 2022). Chen et al. (2024) observed that polybutylene adipate terephthalate (PBAT) microplastics can masquerade as soil carbon to promote the formation of particulate organic carbon, and facilitate the generation of mineral-associated organic carbon and dissolved organic carbon through microbial transformation of PBAT-derived C and selective consumption in dissolved organic nitrogen. To comprehend the impact of microplastics on soil carbon fluxes, further investigation is warranted to explore how both conventional and biodegradable microplastics influence the accumulation and stabilization mechanisms of different carbon fractions. Current studies have suggested divergent effects of microplastics on soil CO_2 emissions, such as increasing (Awet et al. 2018; Shi et al. 2022a; Xiao et al. 2022; Yang et al. 2018), decreasing (Chen et al. 2023b; Wang et al. 2016; Yu et al. 2021), or having no significant effects on CO_2 emissions (Blöcker et al. 2020; Yang et al. 2018). However, whether microplastics directly contribute to CO_2 emissions is still unclear. Liu et al. (2017) reported an accelerated degradation of SOM with high concentrations of PP addition, which led to distinct metabolite profiles after 7 and 30 days. Xiao et al. (2021) carried out a three-source-partitioning study and showed that low microplastic application can more significantly accelerate SOM decomposition than high microplastic application. However, Yu et al. (2021) showed that microplastic addition reduced the decomposition of SOM by decreasing microbially available SOM but increasing mineral-associated organic matter content.

DOM is a more sensitive indicator of soil quality changes in the SOM pool (Gong et al. 2009) and has a central role in numerous physical, chemical, and biological processes in soil (Kalbitz et al. 2000). Although only accounting for < 0.25% of the total SOM, DOM is crucial in regulating the turnover of SOM, the transport of nutrients, the solubility and mobility of heavy metals and organic pollutants, and the activity of microbes (Kalbitz et al. 1997, 2003; Temminghoff et al. 1997). Microplastic input could alter the quantity of DOM in soil, which depends on the imbalance between the production and mineralization of DOM. For example, a majority of studies have observed increased DOM content after the addition of different microplastics (Liu et al. 2017; Meng et al. 2022; Shi et al. 2022a; Zhou et al. 2020a). This is understandable, as the increased activity of enzymes involved in the degradation of recalcitrant (phenolic) compounds may lead to the decomposition of the poorly dissolved large compounds in SOM into easily dissolved small compounds (Keuskamp et al. 2015). However, some studies have also suggested a decrease in DOM content in microplastic-introduced soils (Liu et al. 2019), which was potentially due to the sorption of DOM by microplastics and the degradation of DOM (Chen et al. 2018). The divergence may also be associated with polymer type, microplastic concentration, and incubation time (Ren et al. 2020). Moreover, microplastics may also influence the chemodiversity of soil DOM, such as increasing DOM aromaticity and molecular weight and deepening the DOM humification degree. (Chen et al. 2022a, 2023b; Feng et al. 2022; Li et al. 2022b; Liu et al. 2017). For instance, the introduction of biodegradable microplastics could enhance the relative abundance of labile compounds in soil DOM, such as lipid-like, protein/amino sugar-like, and carbohydrate-like compounds, while conventional polystyrene (PS) microplastics may decrease the relative abundance of stable compounds like lignin-like and more condensed aromatic-like compounds (Sun et al. 2022). As suggested by Qiu et al. (2024), labile components underwent degradation and transformation after microplastic addition, leading to increased aromaticity and oxidation degree, reduced molecular diversity, as well as altered nitrogen and sulfur contents within soil DOM. The changes in soil DOM might be the combined results of microplastic-derived DOM input, organo-organic and organo-mineral interactions, and microbial degradation. These factors warrant further investigation. In fact, chemodiversity is an important indicator to evaluate the environmental reactivity and destination of soil DOM. In general, DOM of lower molecular weight and aromaticity has higher bioavailability (Fouché et al. 2020; Ye et al. 2020). Additionally, DOM contains polar groups and phenolic structures, which have a stronger binding force with heavy metals (Dong et al. 2020; Wang et al. 2018). Therefore, the shifts in DOM compositions after microplastic input are an important topic.

There are several hypothesized pathways by which microplastics affect the turnover of native SOM (Fig. 2). (1) Microplastics could change soil physicochemical properties such as soil aggregates (habitats for SOM stabilization) and therefore affect microbe growth and activity, thus changing the turnover of native SOM. (2) Microplastics could serve as an inert carbon source and establish a unique ecological niche for microbial settlement and growth, thus affecting SOM turnover by directly altering the microbial community. (3) Microplastics can adsorb SOM, interact with soil minerals, and affect the interaction between minerals and SOM, thus affecting SOM accessibility to microbes. (4) Microplastics may impact SOM mineralization through negative priming effects stemming from dilution and substrate switching or positive priming effects arising from co-metabolism, etc. The latter is likely to have marginal importance given the inert nature of microplastic carbon. Alternatively, these effects might be initiated by the more easily metabolizable bioplastics or microplastic leachates (Zhang et al. 2023a). (5) Conventional PE microplastics can influence CO_2 emissions solely by altering DOM electron transfer capabilities. In contrast, the application of biodegradable microplastics (e.g., PLA) can impact gas emissions by increasing both the quantities and transfer capabilities of soil DOM (Shi et al. 2023).

3.4 Microbial response and contribution

Microorganisms have two critical, contrasting roles in controlling terrestrial canbon fluxes: promoting release of canbon to the atmosphere through their catabolic activities, but also preventing release by stabilizing canbon into a form that is not easily decomposed (e.g., the accumulation and stabilization of microbial necromass) (Liang et al. 2017). It is of significant importance to understand the adaptive and evolutionary responses of soil microorganisms to microplastics.

3.4.1 Microbial community structure

The soil microbial community is an important player in regulating nutrient cycling, maintaining soil structure, and detoxifying noxious chemicals. In addition, microbial diversity serves as a sensitive indicator of soil quality, which can reflect subtle changes and soil function (He et al. 2015; Sebiomo et al. 2011). Microplastics exhibit certain filtration effects on soil biota (Guo et al. 2020), and recent studies have reported shifts in community structure, diversity, and evolutionary consequences of microbes in soil in the presence of microplastics (Han et al. 2024; Lu et al. 2023; Rillig et al. 2019a).

Microplastics have been proven to serve as substrates for microbial colonization and assemblage in soil environments, leading to the formation of a unique environment termed the plastisphere (Rillig et al. 2023). Distinct differences were noted for the microbial communities between microplastic surface and ambient soils. For example, the PE microplastic surface was found to carry more plastic-degrading bacteria and pathogens than the surrounding soils (Huang et al. 2019). A similar colonization of plastic-degrading bacteria on microplastic surfaces was also observed in an e-waste dismantling field (Chai et al. 2020). The selection phenomena of microplastics on soil biota shed light on screening microbes for microplastic biodegradation.

Many studies have also reported alterations in bulk microbial diversity and composition in soils after microplastic addition. For example, the bacterial composition in microplastic-amended soils showed significant variations from the control after 90 days of incubation (Huang et al. 2019). Membranous PE and fibrous PP could raise the alpha diversities of the soil microbial community (Yi et al. 2021). In contrast, plastic film residues sharply decreased the soil microbial community and its diversity (Wang et al. 2016), which was attributed to the negative effects of lipophilic phthalate ester additives on the soil biota by destroying cell membrane fluidity (Xie et al. 2010; Zhou et al. 2005). Additionally, the soil microbial diversity was lowered in the presence of PE and PVC microplastics (Fei et al. 2020). Until now, the effects of microplastics on microbial community structures have remained unclear. The contradictory results may be attributed to different microplastic types, concentrations, and additive contents in soil. In general, high microplastic concentrations may induce a quick response in the soil microbiota, whereas lower microplastic concentrations may have insignificant effects on the microbial communities. For example, the addition of PE, polyethylene terephthalate (PET), and PVC microplastics at rather low concentrations (<1%, w/w) after 9 months barely altered the microbial community structure in soil (Judy et al. 2019).

To ease traditional microplastic contamination, biodegradable microplastics have received wide attention and application. As biodegradable microplastics have short lifespans, they are supposed to have more significant effects on soil ecosystems regarding changes in soil biota, greenhouse gas emissions, plant performance, etc. For example, more significant effects on wheat growth were observed for biodegradable microplastics compared to traditional PE microplastics, potentially attributed to microbial immobilization (Qi et al. 2018). More significant variations in bacterial communities, such as the increase in the relative abundance of the genera *Bacillus* and *Variovorax*, were noted for biodegradable microplastics than for traditional PE microplastics (Qi et al. 2020b). In addition, PES and PP were observed to increase root symbiosis by arbuscular mycorrhizal fungi, whereas PET had the opposite effect (de Souza Machado et al. 2019). Notably, microplastics are N-limited carbon materials. Thereby, Zhang et al. (2023a) proposed that bioplastics may promote the proliferation of fast-growing r-strategists through co-metabolism, thereby fostering a positive priming effect in the short term. Conversely, under conditions of severe nitrogen deficiency and labile carbon exhaustion, this may lead to the active growth of K-strategists alongside microbial necromass from r-strategists.

In general, microplastics possess a range of effects on soil microbial community, which are still largely unknown. To better underline the ecosystem effects of microplastics, the adaptive and evolutionary responses of soil biota to microplastic stresses should be further addressed in future studies (Rillig 2018; Rillig et al. 2019a).

3.4.2 Enzyme activity

Soil enzymes are useful for monitoring soil health because of their sensitivity to soil stress, energy flow, and nutrient availability (Wang et al. 2015b). Microplastics could affect the excretion of various soil enzymes by microbes, and extracellular enzymes could attach to microplastic surfaces or affect other soil substrates, thus regulating microplastic degradation and turnover of SOM. de Souza Machado et al. (2018b) first evaluated the effects of microplastics on the hydrolysis of fluorescein diacetate enzyme activity and observed a significant correlation between microplastic concentration and microbial activity irrespective of polymer type. This is consistent with the increased hydrolysis of fluorescein diacetate enzyme activity after PE addition (Liu et al. 2017). Similarly, the addition of membranous PE, fibrous PP, and microsphere PP all increased the urease, dehydrogenase, and alkaline phosphatase enzyme activities in soil (Yi et al. 2021). Additionally, the addition of PP microplastics at a rather high application rate (28%, w/w) could significantly increase the activities of fluorescein diacetate hydrolase and phenol oxidase in sandy loam soils (Yang et al. 2018), thus affecting soil C, N, and P cycling and increasing nutrient availability to plants by enhancing microbial hydrolytic activity on SOM (Liu et al. 2017; Yang et al. 2018). However, a lower application rate (7%, w/w) of PP only had marginal effects on the enzyme activities of fluorescein diacetate hydrolase, urease, and phenol oxidase (Yang et al. 2018). In addition, PE and PVC microplastics could increase the activity of urease and acid phosphatase, whereas the activity of fluorescein diacetate hydrolase activity was inhibited (Fei et al. 2020). Decreases in dehydrogenase activity and enzyme activities involved in the C-(β-glucosidase and cellobiohydrolase), N-(leucine-aminopeptidase), and P-(alkaline-phosphatase) cycles were also observed in soils after 28 days of incubation under 100 and 1000 ng g^{-1} PS nanoplastic treatments, indicating a broad and detrimental impact of PS nanoplastics on soil microbial activity (Awet et al. 2018). Moreover, the residues of plastic film (67.5 kg ha^{-1}) significantly lowered the activity of fluorescein diacetate hydrolase and dehydrogenase by 10% and 20%, respectively (Wang et al. 2016), which may be attributed to the negative effects of concomitant plastic additives. The contradictory results of enzyme activities after microplastic application are potentially attributed to microplastic concentrations and constitutions, plastic additives, and soil properties, the effects of which should be further addressed to better understand the mechanisms. Song et al. (2023) elucidated that PVC elevated β-glucosidase, leucine aminopeptidase, and acid phosphatase activities in both hot- and coldspots within the rice rhizosphere. In contrast, PLA influenced enzyme activities exclusively in the hotspot soil, showing no impact in the coldspot soil. These variations arose from changes in microbial enzyme systems favoring nutrient mining, potentially mitigating some of the adverse effects of microplastics on soil nutrient processes. The close association between soil enzyme activities and soil carbon dynamics necessitates greater emphasis on investigating the response of enzyme activity to microplastic addition, as well as its correlation with other biotic and abiotic processes in soil.

3.4.3 Microbes and enzymes for microplastic degradation

Microplastics can establish a unique ecological niche for certain microbes by providing habitat for microbial settlement and growth. In return, these microbes may contribute to the degradation of microplastics by utilizing polymer carbon as an inert carbon source, consequently impacting the mineralization process of native SOC. As summarized in Table 1, both bacteria and fungi have been proven to have the potential to promote the degradation of microplastics (Russell Jonathan et al. 2011; Shah et al. 2008; Zafar et al. 2013). Some studies have also shown that several bacteria and fungi can use plastic as the sole carbon source, including either in solid or liquid matrices, such as soil (Mohan et al. 2016), compost (Jeon and Kim 2013), and sea water (Harshvardhan and Jha 2013), thus highlighting the potential of such microbes for plastic remediation. Other microbes involved in plastic degradation include *Alcaligenes faecalis*, *Comamonas acidovorans* TB-35, *Pseudomonas putida*, *Pseudomonas stutzeri*, *Saccharomycopsis*, *Streptomyces* sp., and *Staphylococcus* sp. (Akutsu et al. 1998; Benedict et al. 1983; Caruso 2015; Ghosh et al. 2013).

During the degradation process, it is also very important to identify the enzymes involved (Auta et al. 2018; Jaiswal et al. 2020). Extracellular enzymes secreted by microbes are prone to depolymerize microplastics through hydrolysis reactions (Shah et al. 2008). Lipases, cutinases, carboxylesterases, and laccases have been proven to efficiently degrade microplastics (Lucas et al. 2008). The polymer chain can be cleaved into micromolecular water-soluble intermediates, which may be absorbed by the cells and undergo a special metabolism (Gewert et al. 2015). The final degradation products may end up as CO_2, H_2O and CH_4 and are released into the ambient environment (Tokiwa et al. 2009).

3.4.4 Contribution of microbial biomass and necromass to soil carbon pool

Microbial biomass represents the living or actively growing microorganisms, which is associated with the decomposition efficiency of SOM. To date, data on the effects of microplastics on the accumulation of soil microbial biomass are far from sufficient. A 28-day laboratory incubation showed that the addition of 100 and 1000 ng g^{-1} PS nanoplastics both significantly decreased soil microbial biomass (Awet et al. 2018), suggesting potentially broad antimicrobial activity of PS nanoplastics on soil microbiota. Interestingly, a gradual increase in soil microbial biomass with time was observed for 10 ng g^{-1} PS nanoplastic-amended soil throughout 28 days of incubation (Awet et al. 2018). At day 1, the soil microbial biomass remained almost unchanged, whereas the enzyme activities, basal respiration rate, and metabolic quotient decreased, suggesting a sublethal effect with 10 ng g^{-1} PS nanoplastic amendment. At day 28, the soil microbial biomass was significantly higher, potentially due to the increased antimicrobial activity of PS nanoplastics against some microbial genera over time. Thus, dead cells might provide substrates for resistant microorganisms and thus result in cryptic growth (PostGate 1967). Similarly, several studies also observed a decreased microbial biomass in 1% PP-, PE-, or PLA-amended soils (Blöcker et al. 2020; Shi et al. 2022a). In this case, the decrease in microbial biomass was unlikely caused by toxicity effects, as the used plastics were free of antimicrobial additives. However, Zhang et al. (2023a) noted that bioplastics, such as PHA, PBS, and PLA, elevated microbial biomass carbon and dissolved organic carbon levels. This implies that biodegradable microplastics can concurrently expedite microbial assimilation and the transformation of SOM into dissolved organic substrates.

Microbial necromass refers to the non-living remnants of microorganisms, constituting over 50% of SOC

Table 1 Microbes involved in microplastics biodegradation

Microbes	Enzyme involved	Polymer type	Sample information	Degradation efficiency	Methodology	Other information (e.g., time span and carbon source)	Reference
Bacillus cereus	N/A	PE, PET, and PS	Sediment, Peninsular Malaysia	1.6% (PE), 6.6% (PET), and 7.4% (PS)	N/A	40 d, sole carbon source	(Auta et al. 2017)
Bacillus gottheilii	N/A	PE, PET, PP, and PS	Sediment, Peninsular Malaysia	6.2% (PE), 3.0% (PET), 3.6% (PP), and 5.8% (PS)	N/A	40 d, sole carbon source	(Auta et al. 2017)
Bacillus pumilus M27	N/A	PE	Sea water, Arabian Sea coast, India	1.5%	FTIR and SEM	30 d	(Harshvardhan and Jha 2013)
Bacillus subtilis H1584	N/A	PE	Sea water, Arabian Sea coast, India	1.75%	FTIR and SEM	30 d	(Harshvardhan and Jha 2013)
Bacillus subtilis MZA-7	Esterase	PUR	Soil, dumping area in Islamabad, Pakistan	N/A	FTIR, SEM, and GC–MS	N/A	(Shah et al. 2013)
Bacillus sp. strain 27	N/A	PP	Sediment, Peninsular Malaysia	4%	FTIR and SEM	40 d	(Auta et al. 2018)
Bacillus sp.	Depolymerase enzyme	PS	Soil, dumping area in Thiruvananthapuram, India	23%	FTIR, SEM, TGA, and NMR	30 d, sole carbon source	(Mohan et al. 2016)
Brevibacillus borstelensis	N/A	PE	Soil, disposal site at the polyethylene production plant of Carmel Olefins	11%	FTIR	30 d, 50°C	(Hadad et al. 2005)
Chelatococcus sp. E1	N/A	PE	Compost	N/A	FTIR	N/A	(Jeon and Kim 2013)
Enterobacter sp	Depolymerase enzyme	PE	Degraded plastic waste samples	12%	FTIR, SEM, TGA, and NMR	30 d, 30°C, 150 rpm	(Sekhar et al. 2016)
Ideonella sakaiensis 201-F6	Glycoside hydrolases	PET	Sediment, soil, wastewater, and activated sludge from a PET bottle recycling site	N/A	SEM	N/A	(Yoshida et al. 2016)
Kocuria palustris M16	N/A	PE	Sea water, Arabian Sea coast, India	1%	FTIR and SEM	30 d	(Harshvardhan and Jha 2013)
Pseudomonas sp.	Depolymerase enzyme	PS	Soil, dumping area in Thiruvananthapuram, India	< 10%	FTIR, SEM, TGA, and NMR	30 d, sole carbon source	(Mohan et al. 2016)
Pseudomonas aeruginosa	N/A	LDPE	Nutrient broth Temp	6.5–8.7%	FTIR and SEM	60 days at 37°C on 180 rpm	(Gupta and Devi 2020)
Rhodococcus sp. strain 36	N/A	PP	Sediment, Peninsular Malaysia	6.4%	FTIR and SEM	40 d	(Auta et al. 2018)
Thermobifida fusca	Thermophilic polyester hydrolases	PET	Nutrient rich Temp	> 50%	SEM	96 h, 70°C	(Wei et al. 2019a)
Vibrio	N/A	PET	Nutrient broth Temp	35%	FTIR, SEM, XRD	6 weeks, 37°C	(Sarkhel et al. 2019)
Zalerion maritimum	N/A	PE	N/A	> 43%	FTIR, SEM, and NMR	14 d	(Paço et al. 2017)

[*] Here, "N/A" represents that the information is not provided

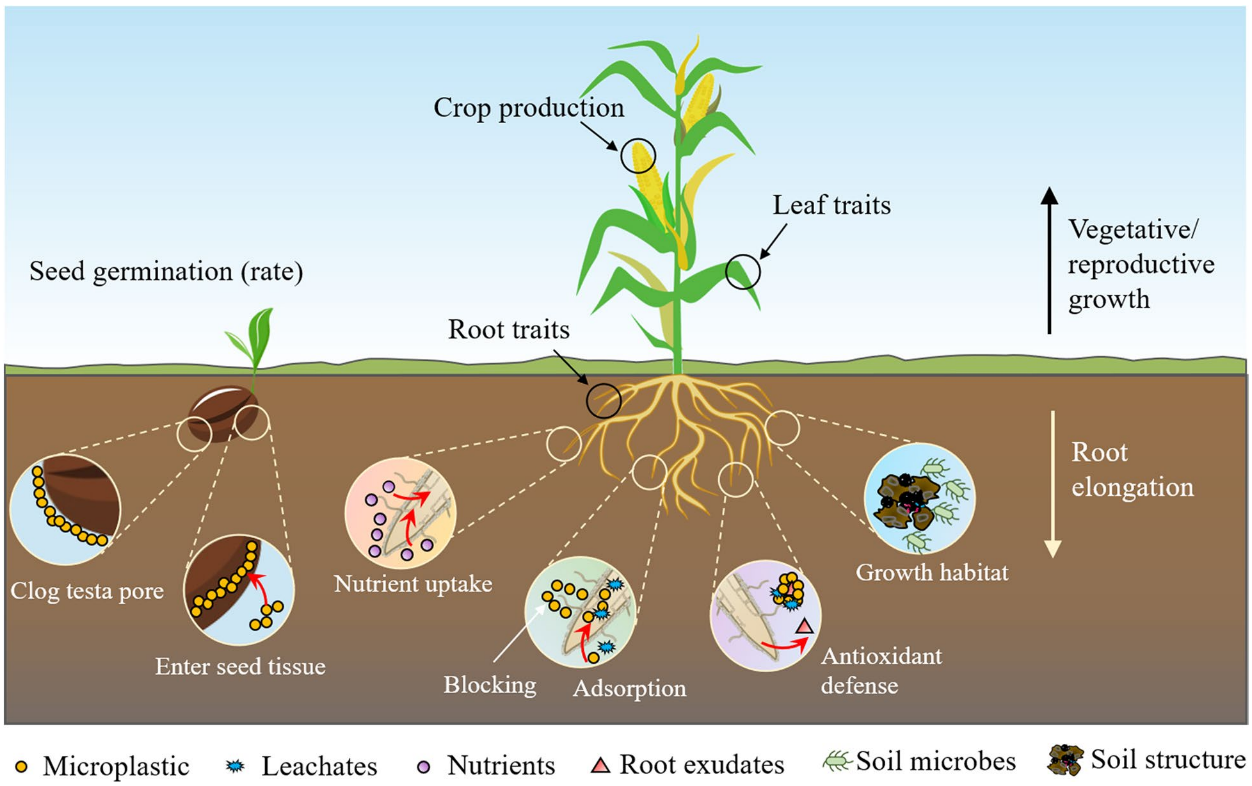


Fig. 3 Concept graphic about the effects of microplastics on plant performance. Microplastics can impede seed germination by obstructing testa pores and infiltrating seed tissues. Furthermore, they can influence aboveground plants' vegetative or reproductive growth. Additionally, microplastics have the potential to alter root traits and rhizosphere, thereby affecting root elongation

pools and approximately 40 times the amount of live microbial biomass carbon (Liang et al. 2017). Chen et al. (2023b) discovered that the addition of PE microplastics could decrease fungal necromass carbon but had minimal impact on bacterial necromass carbon. Zhang et al. (2024) observed distinct effects on microbial necromass accumulation between conventional PE microplastics and biodegradable microplastics (polypropylene carbonate and polybutylene adipate terephthalate synthetic material), with these effects varying across different land-use types.

We propose several hypothesized mechanisms through which microplastics impact the accumulation of microbial necromass carbon. (1) The soluble fraction of microplastics (e.g., plastic additives, long-chain alkanes, high-molecular-weight acids, and benzoic ether) can act as a carbon source for soil microbes, thereby influencing microbial population and necromass accumulation. (2) The toxic compounds present in microplastics may pose a threat to soil microbes, affecting their growth and death pathways. (3) Microbial necromass could serve as a carbon source and enhance the availability of soil carbon (Caldwell 2005). Microplastic-derived carbon may have positive or negative priming effects on native SOC mineralization, including the mineralization of microbial necromass carbon. (4) As soil aggregates are fundamental units for SOC protection, microplastics can also mediate microbial necromass accumulation and decomposition by impacting soil aggregation. Changes in soil aggregation also shape microbial habitats and influence microbial population dynamics and mortality rates. Moreover, binding agents secreted by soil microbes could reciprocally alter soil aggregation processes that subsequently affect microbial metabolism and SOM turnover. Currently, there is limited research on the accumulation of microbial necromass caused by microplastics; thus, further investigation is required to verify the underlying mechanisms.

4 Effects of microplastics on terrestrial plants

Plants play a vital role in maintaining the health of soil ecosystems and are widely applied to detect and evaluate the toxicity of different environmental stressors. Additionally, plants significantly contribute to soil carbon sequestration through photosynthetic carbon fixation. Moreover, the plant rhizosphere serves as a hotspot for microbial activity and exerts substantial influence on the soil carbon cycle. The presence of microplastics can

Table 2 Effects of microplastics on plant performance

Species	Polymer	Shape	Size (μm)	Concentration	Incubation time	Endpoint	Major hypothesized effect pathway	Reference
Allium cepa	PS	Sphere	0.1	25, 50, 100, 200, 400 mg/L	24, 48, 72 h	Root length (–) Oxidative stress Gene expression	Reactive oxygen species production Chromosomal abnormalities Inhibition of cell cycle regulators	(Maity et al. 2020)
Allium fistulosum	PES	Fiber	8 × 5000	0.2% (fresh weight)	~ 1.5 months	Root biomass (+) Root-leaf ratio (+/–) Root length (+) Root average diameter (–) Total root area (+) Root tissue density (+/–) Root colonization by AMF (+/–) Leaf Traits Total biomass (+)	Soil nitrogen availability Soil structure and composition Water transport Microbial activity	(de Souza Machado et al. 2019)
	PA	Bead	15–20	2.0%, w/w (fresh weight)				
	HDPE	Fragment	643	2.0%, w/w (fresh weight)				
	PET	Fragment	187	2.0%, w/w (fresh weight)				
	PP	Fragment	624	2.0%, w/w (fresh weight)				
	PS	Fragment	492	2.0%, w/w (fresh weight)				
Gossypium hirsutum	Mulching film residue	N/A	N/A	Varied film residue density	N/A	Boll weight (–) Yield (–) Total biomass (–)	N/A	(Dong et al. 2015)
Lepidium sativum	Bioplastic	Film	N/A	12.5 g kg^{-1}	72 h	Seed germination (N/A)	N/A	(Sforzini et al. 2016)
	PS	Fragment	0.05, 0.50, 4.80	10^3–10^7 items mL^{-1}	4, 8, 24, 48, 72 h	Seed germination (–)	Physical blockage of the pores in the seed capsule	(Bosker et al. 2019)
	PS	Fragment	0.05, 0.50, 4.80	10^3–10^7 items mL^{-1}	4, 8, 24, 48, 72 h	Root growth (+/–)	Accumulation on the root hairs	(Bosker et al. 2019)
	PP, PE, PVC, PE + PVC	Fragment	< 125	0.02%, w/w	6, 24 d	Day 6: Seed germination (–) Shoots height (–) Leaf number (N/A) Shoots biomass (+) Oxidative stress (+) Day 21: Seed germination (–) Shoots height (N/A) Leaf number (N/A) Shoots biomass (–) Oxidative stress (+) Chlorophyll (+) Carotenoid (–)	N/A	(Pignattelli et al. 2020)

Table 2 (continued)

Species	Polymer	Shape	Size (μm)	Concentration	Incubation time	Endpoint	Major hypothesized effect pathway	Reference
Lolium perenne	HDPE	N/A	0.48–316	0.1%, w/w	30 d	Seed germination (N/A) Shoot length (N/A) Shoot biomass (N/A) Root biomass (+) Chlorophyll (N/A)	N/A	(Boots et al. 2019)
	PLA	N/A	0.6–363	0.1%, w/w	30 d	Seed germination (−) Shoot length (−) Shoot biomass (N/A) Root biomass (N/A) Chlorophyll (N/A)	Immobilization of nutrients by organic compounds originating from degradation	(Boots et al. 2019)
	Synthetic fiber	Fiber	N/A	0.001%, w/w	30 d	Seed germination (−) Shoot length (N/A) Shoot biomass (N/A) Root biomass (N/A) Chlorophyll (N/A)	N/A	(Boots et al. 2019)
Lycopersicon esculentum L	PS	Fragment	52.48 ± 20.93, 368.13 ± 127.11	0, 10, 100, 500, 1000 mg/L	7 d	seed germination (−) Root length (−) Root weight (fresh) (−) Oxidative stress	Microplastics reduced the activities of SOD, POD, and CAT as well as the contents of SS and SP	(Shi et al. 2022b)
	PE	Fragment	75.37 ± 17.55, 241.97 ± 81.55			seed germination (−) Root length (−) Root weight (fresh) (−) Oxidative stress		
	PP	Fragment	88.11 ± 28.53, 273.52 ± 111.69			seed germination (−) Root length (N/A) Root weight (fresh) (N/A) Oxidative stress		
Oryza sativa L	PE, PAN, PET	Granule	10 and 200	0.05%, w/w	5 months	Plant height (N/A) Tillers number (N/A) Leaf SPAD (+) NDVI (N/A) Rice grain yield (+)	N uptake capacities of rice grain	(Chen et al. 2022b)

Table 2 (continued)

Species	Polymer	Shape	Size (μm)	Concentration	Incubation time	Endpoint	Major hypothesized effect pathway	Reference
Oryza sativa L	PVC, PLA	N/A	155–180	10%, w/w	30 d	Shoot length (–) Shoot biomass (–) Root length (–) Root biomass (–)	Microplastics strengthened the co-occurrence networks among bacteria, increased microbial functionality resilience, and enhanced competition with neighboring roots for nutrient mining	(Song et al. 2023)
Sorghum saccharatum	Bioplastic	Film	N/A	12.5 g kg^{-1}	72 h	Seed germination (N/A)	N/A	(Sforzini et al. 2016)
Triticum aestivum	Bioplastic > LDPE	Film	50–1000	1%, w/w	4 months	Vegetative and reproductive growth (–) Chlorophyll (N/A)	N/A	(Qi et al. 2018)
Vicia faba	PS	Sphere	0.1, 5	10, 50, 100 mg L^{-1}	48 h	Root length (–) Fresh/dry weight (–) Antioxidant enzymes activities (–) Genotoxicity	Physical blockage of cell connections or cell wall pores and disruption of nutrients transport	(Jiang et al. 2019)

"N/A" represents no significant effects or not provided, "+" represents positive effects, and "–" represents negative effects. "N/A" represents the information is not provided impact both rhizodeposit secretion by plants and the decomposition and stabilization of rhizodeposits within the soil carbon pool, thereby bearing significant implications for the soil carbon cycle.

4.1 Effects of microplastics on plant performance

As summarized in Fig. 3 and Table 2, microplastics have been observed to affect plant performance regarding seed germination (Bosker et al. 2019; Sforzini et al. 2016; Shi et al. 2022b), vegetative and reproductive growth (Miao et al. 2023; Qi et al. 2018), root and leaf traits (de Souza Machado et al. 2019), crop production (Chen et al. 2022b; Hegan et al. 2015), and ecotoxicity and genotoxicity on plants (Jiang et al. 2019).

The incubation time could affect the apparent impact of microplastics. For example, PS microplastics within a size range of 0.05–4.80 µm were observed to delay seed germination of *Lepidium sativum* during 8 h of incubation, whereas the inhibition effects were no longer observed after 24 h of incubation (Bosker et al. 2019). Additionally, the promotion or inhibition effects on root growth were no longer found as the incubation time was prolonged from 24 to 48 h (Bosker et al. 2019). Moreover, different impacts of microplastics were noted in regard to the physiological responses of *Lepidium sativum* after 6 days of acute exposure and 21 days of chronic exposure experiments (Pignattelli et al. 2020). Therefore, the design of the experimental span could affect the assessment of the ecological risks of microplastics.

Microplastic concentration is also a crucial factor governing plant performance. For instance, microplastics at higher concentrations exerted more significant delaying effects on seed germination (Bosker et al. 2019). Additionally, PS microplastics at higher concentrations exerted more significant inhibitory effects on the root length and fresh and dry weight of *Vicia faba* (Jiang et al. 2019). In addition, some research data indicated that crop yields could decrease when mulch film residues amounted to 58.5 kg ha^{-1} (Dong et al. 2015; Li et al. 2014).

Different polymer types may exert different effects on the exposed plants. For example, PE is widely applied as mulch film in agriculture, and residues have been observed to decrease crop production (Jiang et al. 2017; Zhang et al. 2016, 2023b). However, bioplastic film, a seemingly more environmentally friendly alternative to traditional mulch film, was observed to exert a stronger negative impact on the vegetative and reproductive growth of *Triticum aestivum* (Qi et al. 2018), which aroused debates about its wide application (Campanale et al. 2024; Tan et al. 2016). In addition, the impact on the root-leaf ratio, root tissue density, root colonization by arbuscular mycorrhizal fungi (AMF), and leaf traits of *Allium fistulosum* varied with polymer type (de Souza Machado et al. 2019).

Microplastic size and shape also have very different effects on plant performance. For example, PS microplastics were prone to inhibit the growth of *Vicia faba*, whereas stronger inhibiting effects were observed for PS nanoplastics; PS microplastics of larger size exerted stronger delaying effects on seed germination (Bosker et al. 2019). Additionally, exposure to 50 nm PS particles resulted in a significant increase in root growth of *Lepidium sativum* after 24 h of incubation, while exposure to 500 nm PS particles led to a significant decrease in root growth, and exposure to 4800 nm had no significant influence (Bosker et al. 2019). Rillig et al. (2019b) proposed the major hypothesized effect pathway for different shapes and sizes and suggested that fibers of large size may exert positive effects on plant growth by altering soil structure and bulk density, films of intermediate size may exert negative effects on plant growth by increasing soil water evaporation, and beads or fragments of minor size could also possess certain effects similar to minor changes in soil texture.

In the aforementioned studies, the effects of shape or size cannot be completely separated from those of polymer type. Considering the various impacts of microplastics owing to their dazzling range of chemical makeup, additives, persistence, surface properties, sizes and shapes, future research should investigate the role and correlation of different factors on root growth parameters, nutrient uptake, and root-colonizing microbes over different time spans (Rillig et al. 2019b).

4.2 Mechanisms underlying the impact of microplastics on plant performance

Different mechanisms were noted for the impact of microplastics on plant performance, such as serving as root barriers, changing soil microbial structure and metabolism, accumulating in plants, regulating the response of plants to microplastic contamination, etc. Such effects could occur separately or concurrently but are discussed separately as follows (Fig. 3).

4.2.1 External effects of altering the environment in regard to plant growth

First, microplastics could alter soil physicochemical parameters, which may alter water transport, nutrient availability, microbial activities, and penetration resistance for plant roots, thus affecting plant performance. For instance, PES exerted stronger effects on soil structure and interaction with water than other polymer types, which potentially drove the response of plant traits (de Souza Machado et al. 2019). The substantially enhanced water-holding capacity in soil and the maintenance of high water levels in plant tissues by PES addition could alter plant physiological status (e.g., physiological proxies of photosynthesis) and thus result in changes in specific plant traits (Faucon et al. 2017). Song et al. (2023) also suggested that PVC could suppress shoot growth by influencing nutrient and water absorption abilities (Qi et al. 2020a), as well as integrating into soil aggregates to inhibit the activities of soil enzymes and change the uptake of nutrients and water (Gao et al. 2021; Hou et al. 2021; Ng et al. 2021). Alternatively, shifts in water dynamics may affect nutrient availability by altering chemical speciation processes within soils or changing soil microbial activities (de Souza Machado et al. 2019). As plant performance is also highly dependent on soil biota (Wagg et al. 2014), and particularly on root-colonizing microbes such as N-fixers (Liu et al. 2024), pathogens and mycorrhizal fungi (Powell and Rillig 2018), the altered microbial community either by the direct impact of microplastics or the indirect impact of changed soil physicochemical properties could also play a significant role. For example, the changed habitat due to PES addition resulted in the root colonization of AMF, which conferred positive effects to plant growth by increasing nutrient availability (de Souza Machado et al. 2019).

Second, microplastics can accumulate at plant roots and affect root growth and nutrient uptake. For example, PS microplastics could affect root growth of *Lepidium sativum* by accumulating on the root hairs (Bosker et al. 2019). Similarly, PS microplastics 5 μm in size, due to their large size, were prone to accumulate at the root surface and thus affected the root growth and fresh and dry weight of *Vicia faba*, potentially by blocking the absorption or uptake of essential nutrients and water through root tips (Jiang et al. 2019).

Third, microplastics contain some biogeochemically active elements that might exert selective pressure on soil microbes and plants. For nitrogen-free microplastics like PLA, they can be hydrolyzed to form water-soluble low-molecular-weight oligomers, which induce microbial immobilization and assimilation of essential nutrients and increase stress in plants (Song et al. 2023). For N-containing microplastics (i.e., PA, polyacrylonitrile, and polyaramide), their effects on soil structure and plant performance could be partly attributed to the enrichment of soil nitrogen. PA is generally manufactured through the polymerization of amines and carboxylic acids, and its components loosely interact (Palmer 2001). Thus, the remaining monomers and compounds from the production process could easily leach into the soil and lead to effects analogous to fertilization. This can partly explain the nearly twofold increase in leaf N content in *Allium fistulosum* by PA application (de Souza Machado et al. 2019). For polymers containing chloride, such as PVC, the leaching of certain components could lead to biogeochemical changes, as very strong filtration effects on microbes were observed for PVC-derived DOM (Zhang et al. 2022). The phenomenon in which plants are unable to counteract PVC toxicity and produce high concentrations of H_2O_2 during the time period of chronic exposure is also potentially related to the chemical toxicity of chloride (Pignattelli et al. 2020). Moreover, it is plausible to speculate that in the long run, even the inert carbon in plastic polymers may constitute a relevant carbon pool in the soil (Rillig et al. 2019a) and a selection pressure for soil microbes (Chen et al. 2022d). Particularly for bioplastics with weaker persistence, the biodegradation and microbial immobilization of polymer carbon could be more profound, which may lead to more pronounced effects on plant performance (Qi et al. 2018).

Fourth, microplastics may carry toxic additives or adsorb pollutants such as the concurrently used pesticides (Ramos et al. 2015), which may either ease or exacerbate the environmental stress from microplastics (Zhou et al. 2020b). Microplastics can increase the exposure of soil organisms to adsorbed pollutants (Hodson et al. 2017), and the adsorbed toxic substances can exert adverse effects on plant roots and their symbionts (Gaylor et al. 2013; Wei et al. 2019b), thus negatively affecting plant growth. In addition, microplastics may have stimulated opportunistic plant pathogens and induce phytotoxicity due to acidification of the soil during PLA degradation (Song et al. 2023). On the other hand, the adsorption of contaminants by microplastics could lower their availability to soil biota and plants, thus leading to positive effects on plant growth (Kleinteich et al. 2018; Rehse et al. 2018). Thus far, the combined effects of microplastics and their coexisting pollutants are still largely unknown and merit further investigation (Wang et al. 2019).

4.2.2 Internal effects through uptake of microplastics by plants

The root rhizodermis is a barrier to microplastic uptake. As microsized particles are not expected to be transported into the root, the following discussion mainly concerns nanoplastics, which belong to a broad range of microplastics. Nanoplastics are likely to be transported to and accumulate in terrestrial plants, which may cause damage to plants (e.g., changes in the cell membrane, intracellular molecules, and generation of oxidative stress) (Navarro et al. 2008) and could be a threat to the ecological environment and human health (Maity and Pramanick 2020). For example, Jiang et al. (2019) showed that 100 mg L^{-1} PS nanoplastics (100 nm) exerted a profound inhibitory effect on *Vicia faba* growth potentially by blocking cell connections or cell wall pores thus inhibiting nutrient transport. Additionally, PS nanoplastics could accumulate in the testa pores and delay seed germination of *Lepidium sativum* by physically blocking the pores in the seed capsule (Bosker et al. 2019). However, the realization of nanoplastic toxicity is not limited to entering the plant tissue. Alternatively, plastic leachates, such as additives, plasticizers and flame retardants, may serve as contributors to the toxic effects on plants. An exposure experiment of PS nanoplastics showed morpho- and cytogenotoxic effects in *Allium cepa* by inducing reactive oxygen species production and chromosomal abnormalities and lowering the expression of the *cdc2* gene, whereas no deposition of PS beads inside root tissue was observed (Maity et al. 2020). Microplastics of 4.8 μm in size clogged pores in seed capsules and their adherence to seedlings rather than entering the plant tissue also indicated clogging (Bosker et al. 2019). Microplastics are good vectors for other pollutants, such as heavy metals, dioxins, and persistent organic pollutants (Kumar et al. 2020; Wang et al. 2020). Additionally, microplastics usually contain large varieties of additives of great ecological risks, such as flame retardants, plasticizers, heat stabilizers, and antioxidants (Rillig and Lehmann 2020; Zhou et al. 2020b). Such characteristics could contribute to the negative effects of microplastics on plants.

4.3 Response of plants to microplastic threats

In contaminated fields, plants have developed several strategies to deal with xenobiotics, such as avoiding the intake of such compounds, expanding the root system (Boots et al. 2019), and developing detoxification mechanisms through enzymatic reactions such as oxidation, reduction, and conjugation reactions (Wang et al. 2019). As mentioned above, microplastics can affect the water and nutrient supply to plant roots. Correspondingly, several detoxification mechanisms are involved in alleviating such adverse effects. Long-term stressed plants are prone to secrete extracellular enzymes to degrade microplastics and coexisting pollutants and additives. For example, exposure to PS microplastics leads to shifts in the antioxidant defenses of *Vicia faba* by secreting antioxidant enzymes such as superoxide dismutase and peroxidase enzymes (Jiang et al. 2019). The increased production of H_2O_2 in plants, as well as the triggered production of low molecular weight compounds with antioxidant action, also supported the antioxidant defenses of plants to microplastic stressors (Pignattelli et al. 2020). Moreover, the microorganisms and microanimals in plant roots could alleviate the toxic effects by degrading and utilizing microplastics and coexisting intermediates. Unfortunately, the influence of microplastics on plant health and microplastic biodegradation in the rhizosphere of plants is still largely unknown and merits further study.

5 Future research directions

Due to the irresistible increase in plastic production and the lack of effective waste disposal countermeasures, the current pressure of microplastic contamination in soil is expected to continue for many years to come. To better understand the ecosystem effects and risks of microplastics in the soil environment, there are still some critical unknowns that need to be addressed in future work: (1) elucidating the correlation between alterations induced by microplastics in physicochemical properties and soil carbon cycling; (2) clarifying the influence of microplastics on microbial stabilization and mineral protection of soil carbon, with particular attention given to microbial necromass accumulation and carbon storage within mineral-associated fractions; (3) determining the source of carbon emissions from native SOC and microplastics through utilization of ^{13}C isotope technology; (4) investigating the effects of microplastics on rhizosphere dynamics, particularly microbial activity and function, as well as microbial stabilization and mineral protection mechanisms for rhizodeposits within soils; (5) exploring ecosystem-level consequences associated with so-called "eco-friendly" bioplastics, since microbioplastics may exert more pronounced effects on soil biophysical properties, which should be considered during their safe management within agricultural contexts.

Abbreviations

AMF	Arbuscular mycorrhizal fungi
DOM	Dissolved organic matter
HDPE	High density polyethylene
PA	Polyamide
PBAT	Polybutylene adipate terephthalate
PBS	Polybutylene succinate
PE	Polyethylene
PES	Polyester
PET	Polyethylene terephthalate
PHA	Poly-hydroxyalkanoates
PLA	Polylactic acid
PP	Polypropylene
PS	Polystyrene
PVC	Polyvinyl chloride
SOC	Soil organic carbon
SOM	Soil organic matter

Supplementary Information

The online version contains supplementary material available at https://doi.org/10.1007/s44246-024-00124-1.

Supplementary Material 1.

Authors' contributions

All authors contributed to the study conception and design. Yalan Chen: Conceptualization; Data curation; Validation; Visualization; Writing—original draft. Ke Sun: Conceptualization, Supervision, Funding acquisition, Project

administration, Writing—review & editing. Yang Li, Xinru Liang, Siyuan Lu, Jiaqi Ren, Yuqin Zhang, Zichen Han: Conceptualization, Writing—review & editing. Bo Gao: Supervision, Writing—review & editing. All authors read and approved the final manuscript.

Funding
This work was financially supported by the National Science Foundation for Distinguished Young Scholars (42125703) and National Natural Science Foundation (41977299).

Availability of data and materials
The datasets used or analyzed during the current study are available from the corresponding author on reasonable request. The supporting information contains figure of global microplastics abundance in different soils expressed as mass concentration, and tables of the density information of commonly used microplastics and basic information about density solution used for microplastic extraction.

Declarations

Competing interests
Ke Sun is an editorial board member for Carbon Research and was not involved in the editorial review, or the decision to publish, this article. All authors declare that there are no competing interests.

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Received: 18 November 2023 Revised: 23 March 2024 Accepted: 27 March 2024
Published online: 19 April 2024

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