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Effects of Microplastics on the Terrestrial Environment: A Critical Review

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Highlights

- Microplastics (MPs) cause adverse effects on aquatic and terrestrial environments.
- Various interactions of MPs with soil quality and ecotoxicological impacts need to be further studied.
- Management of MPs are essential to achieve United Nations Sustainable Development Goals.
- Development of standardized methods for analyzing microplastics in soil-plant system is urgently needed.

Abstract

Microplastics are emerging contaminants and there has been growing concern regarding their impacts on aquatic and terrestrial environments. This review provides a comprehensive overview of the current knowledge regarding the sources, occurrences, fates, and risks associated with microplastic contamination in terrestrial environments. This contamination occurs via multiple sources, including primary microplastics (including synthetic materials) and secondary microplastics (derived from the breakdown of larger plastic particles). Microplastic contamination can have both beneficial and detrimental effects on soil properties. Additionally, microplastics have been shown to interact with a wide array of contaminants, including pesticides, persistent organic pollutants, heavy metals, and antibiotics, and may act as a vector for contaminant transfer in terrestrial environments. Microplastics and their associated chemicals can be transferred through food webs and may accumulate across multiple trophic levels, resulting in potential detrimental health effects for humans and other organisms. Although several studies have focused on the occurrence and impacts of microplastic contamination in marine environments, their sources, fate, transport, and effects in terrestrial environments are less studied and not well understood. Therefore, further research focusing on the fate, transport, and impacts of microplastics in relation to soil properties, polymer composition and forms, and land-use types is needed. The development of standardized and harmonized methods for analyzing microplastics in soil-plant ecosystems is essential. Future work should also consider the many interactions of microplastics with soil quality and ecotoxicological impacts on biota in the context of global environmental change.

Keywords: contaminants, microplastics, toxicity, soil properties, trophic transfer

List of abbreviations

1,2-bis(2,4,6-tribromophenoxy)ethane (BTBPE)

Carbon-to-nitrogen (C:N)

Dichlorodiphenyltrichloroethane (DDT)

Dissolved organic matter (DOM)

High-density polyethylene (HDPE)

Life-cycle assessment (LCA)

Low-density polyethylene (LDPE)

Non-polyester microfiber (NPMF)

Organophosphorus esters (OPEs)

Phthalic acid esters (PAEs)

Polyamide (PA)

Polybrominated diphenyl ether (PBDE)

Polychlorinated biphenyls (PCBs)

Polycyclic aromatic hydrocarbons (PAHs)

Polyester (PES)

Polyester microfiber (PMF)

Polyethylene terephthalate (PET)

Polylactic acid (PLA)

Polypropylene (PP)

Polystyrene (PS)

Polyvinyl chloride (PVC)

Ultraviolet (UV)

55 Sustainable Development Goals (SDG)

1. Introduction

Mass production of plastics began in the 1940s with a plethora of different types being manufactured (Mitrano and Wohlleben, 2020). Plastic products have gained popularity worldwide due to their ease of use, durability, and cost-effectiveness (Wijesekara et al., 2018). Widespread and continuous plastic use across many societal sectors, coupled with their long-lasting nature and low degradability, has caused extensive contamination of the natural environment with plastic debris (Rillig, 2012). Carpenter (1972) first revealed the presence of plastic particles in the ocean and documented their potential impacts in the 1970s (Mitrano and Wohlleben, 2020). Renewed scientific interest in microplastics over the past decade has demonstrated that they are present in both aquatic and terrestrial environments, while being an emerging threat to ecosystem functions (Guo et al., 2020). Landfills, urban areas, and beaches are the most severely contaminated areas, with agricultural ecosystems also being significantly affected (Ng et al., 2018; Wang et al., 2019). Recently, there have been increased concerns regarding microplastic contamination of marine ecosystems (Rochman, 2004). However, most marine microplastic pollution is derived from land, from which 4.8–12.7 metric tons of plastics are transferred annually to marine ecosystems (Haward, 2018).

Depending on their size, plastic particles are classified as macro- (>5 mm) (Wijesekara et al., 2018), micro- (1–5 mm), or nanoplastics (< 1 μm) (Bradney et al., 2019a). Microplastic accumulation in terrestrial environments can occur through primary sources by the direct addition of microplastics that are manufactured as micrometer-sized particles in multiple industries (Horton et al., 2017), or secondarily by physical, chemical, and biological fragmentation of macroplastics (de Souza Machado et al., 2018; Horton et al., 2017).

Microplastic contamination in soil can occur through multiple sources, including soil amendments such as biosolids and composts (Bradney et al., 2019a), plastic mulching films (Boots et al., 2019), materials used in greenhouses, irrigation tools (Bläsing and Amelung, 2018), municipal solid waste (Galafassi et al., 2019), wastewater treatment plants (Galafassi et al., 2019), tire wear (Kumar et al., 2020), and atmospheric inputs (Dris et al., 2016). It has been estimated that 63,000 to 430,000 tons of microplastics are released annually to farmlands in Europe, whereas 44,000 to 300,000 tons are released to farmlands in North America annually through sewage sludge application (Nizzetto et al., 2016). China alone utilized over 1.4 million tons of agricultural plastic mulching film in 2017 (Gao et al., 2019). At present, ubiquitous microplastic contamination in all ecosystem types has become a major global concern (Bank and Hansson, 2019).

Despite the short-term benefits of using plastic materials, there is now a growing focus on their long-term effects on soil quality and crop productivity (Qi et al., 2018; Boots et al., 2019). Microplastics interact with and adsorb inorganic pollutants, including trace elements (Bradney et al., 2019a) and organic contaminants such as polychlorinated biphenyls (PCBs), dioxins, dichlorodiphenyltrichloroethane (DDT), and polycyclic aromatic hydrocarbons (PAHs) (Nizzetto et al., 2016). Because of their surface interactions, microplastics can serve as a vector for contaminants to be transported within and between different ecosystem compartments (Bank and Hansson, 2019). Apart from the more obvious environmental concerns related to microplastic pollution, there is also a risk of food chain contamination and human health consequences due to terrestrial plastic contamination (Bradney et al., 2019a). Although several papers have been published recently on different aspects of microplastics in terrestrial environments, to date there is no holistic ecotoxicological overview focusing on the interrelationships of microplastic pollution in plants, micro- and macro-organisms in soil, and their effects on higher animals including humans. This review summarizes the potential sources of microplastic contamination in terrestrial environments, their fates, their effects on soil quality, and their ecotoxicological impacts on plants and other biota.

2. Occurrence and fates of terrestrial microplastics

Microplastics can enter terrestrial ecosystems either directly as primary microplastics, or indirectly as secondary microplastics (Waldman and Rillig, 2020). Primary microplastics are manufactured for specific purposes and products including cosmetics, medical applications, waterborne paints, adhesives, coatings, and electronics. A substantial proportion of primary microplastics enter terrestrial environments through atmospheric deposition (Allen et al., 2019; Brahney et al., 2020; Zhang et al., 2020; Evangeliou et al., 2020). In addition, personal care products, household items, landfills, and application of sludge to agricultural lands also contribute to primary microplastic deposition in terrestrial ecosystems (Karbalaei et al., 2018).

Secondary microplastics (Waldman and Rillig, 2020) are formed by the fragmentation of larger plastics into smaller pieces through exposure to ultraviolet (UV) radiation, wind, tillage, and biological activities, as well as through chemical and mechanical breakdown (Guo et al., 2020; Karbalaei et al., 2018). Plastic mulches, greenhouse materials, soil amendments, irrigation water, municipal solid waste, atmospheric inputs, indiscriminate disposal in landfills, and littering are sources of secondary microplastics in terrestrial environments (Fig. 1) (Galafassi et al., 2019; Bläsing and Amelung, 2018; Bradney et al., 2019b). Nizzetto et al. (2016) reported that the annual additions of microplastics to agricultural soils in Europe through the application of sewage sludge and processed biosolids were approximately 125 and 850 tons per million inhabitants, respectively. Based on the total sludge production in China, Li et al. (2018)

estimated that the average annual sludge-based environmental microplastic addition was 1.56 × 10^{14} particles in China. Plastic mulch films, with thicknesses of 6–20 μm, have been widely used in intensive cultivation systems (Ng et al., 2018), contaminating the soil with their residues in the field (Steinmetz et al., 2016). Dan et al. (2016) observed a significant positive correlation between soil microplastic accumulation and numbers of mulching years. Moreover, microplastics can be transported long distances by wind from surfaces such as landfills and roads, resulting in atmospheric inputs to terrestrial environments (Bläsing and Amelung, 2018; Rillig, 2012).

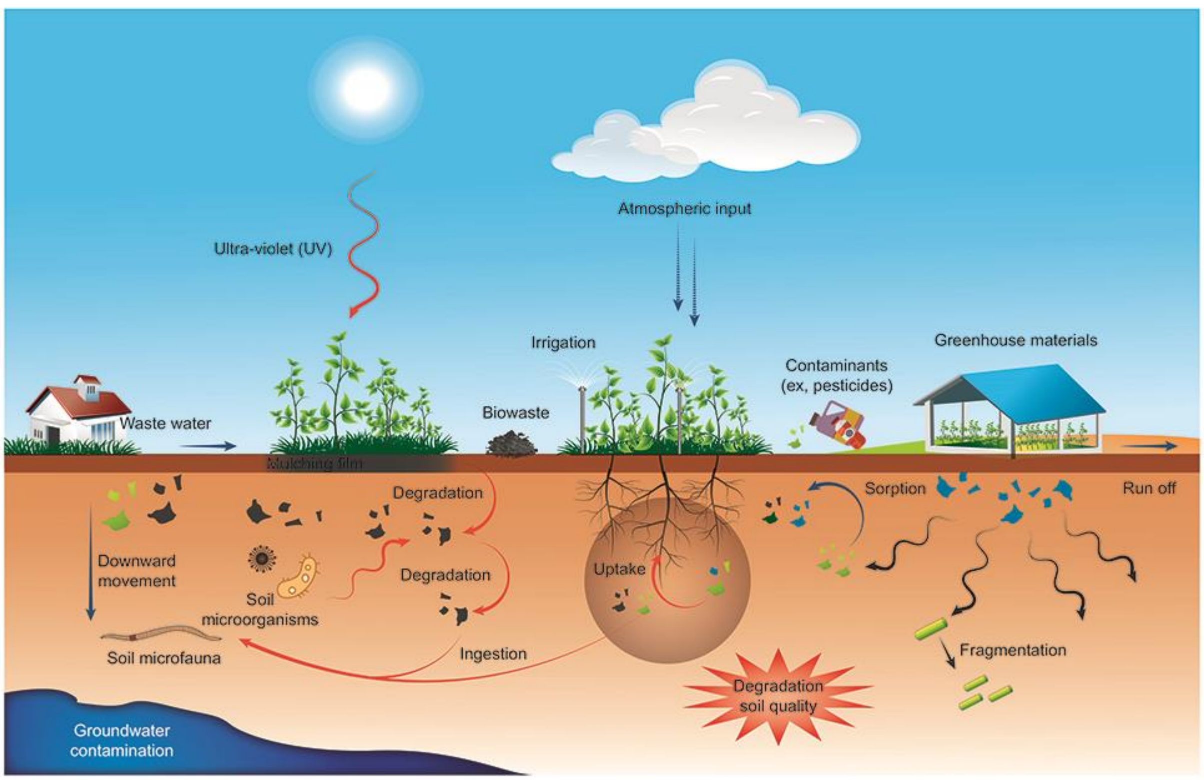

Fig. 1. Sources and fates of microplastics in the terrestrial environment (modified from Rillig and Lehmann, 2020; Kumar et al., 2020).

Plastic particles that enter the soil surface are incorporated into deep soil layers through tillage, animal activities such as ingestion and egestion by earthworms, or water infiltration caused by digging (Rillig et al., 2017; Bläsing and Amelung, 2018; Guo et al., 2020). Microplastics can be degraded by UV radiation (Sen and Raut, 2015), physical abrasion (Zhu et al., 2019), thermal oxidation (Benítez et al., 2013), microbial action (Krueger et al., 2015), or interaction with soil colloids (Ren et al., 2021). The combined effects of these factors can cause the aging of microplastics; however, each factor operates differently during different times, seasons, regions, and conditions. For example, light irradiation plays a vital role during the day, whereas the impact of this factor could be negligible at night (Liu et al., 2021). Moreover, a more significant effect of temperature on the aging of microplastics might be observed in desert regions in summer than in polar regions in winter (Liu et al., 2021). During aging, the physicochemical properties of microplastics, including color, crystallinity, chemical composition, and surface chemistry are altered (Ren et al., 2021). In addition, multiple chemicals including phthalates, retardants, stabilizers, pigments, oligomers, and oxygenated products (e.g., phenols, acetophenones, and carboxylated products) can be released into soil during the aging of microplastics (Liu et al., 2021).

However, aging of microplastics occurs slowly. For example, synthetic fibers in soil samples from USA, collected at depths of up to 100 cm, were identified up to 15 years after sludge application, indicating a long legacy (Zubris and Richards, 2005). The biogenic transport of microplastics in soil may lead to groundwater pollution, uptake by terrestrial plants, and entry into terrestrial food webs (Lwanga et al., 2017). Moreover, runoff of microplastics can be a direct pathway for land-based microplastic contamination of freshwater ecosystems (Ding et al., 2019; Liu et al., 2019), eventually contributing to marine microplastic pollution (Rochman, 2004).

3. Effects of microplastics on soil properties

3.1 *Effects on soil physical properties*

When microplastics are deposited on soil surfaces, they can be immediately incorporated into the soil matrix by external forces, including bioturbation and human activities. This incorporation can trigger changes in soil structure and texture. In highly polluted soils, visual changes can be observed in soil structure and composition, largely because commercial polymers tend to be less dense than common soil particles. Polymer types, forms, and shapes are important features that can be used to evaluate shifts in soil physical properties due to plastic pollution (Lehmann et al., 2021; Wang et al., 2021). Linear-shaped polymers can blend more homogeneously into the soil matrix than non-linear polymers, enhancing soil clumping (Rillig et al., 2017a).

The addition of polyester (PES) fibers (0.1, 0.2, and 0.3 %) to soil was associated with concentration-dependent shifts in soil bulk density, water holding capacity, and water-stable aggregates (Table 1) (De Souza Machado et al., 2018). The bulk densities of soils contaminated with high-density polyethylene (HDPE) fragments, PES fibers, polyethylene terephthalate (PET), polypropylene (PP), and polystyrene (PS) decreased. The formation of water-stable aggregates significantly decreased in PS- and PES-contaminated soils. In contrast, a study conducted with 0, 0.1, and 0.3 % of soil weight of PES microfibers had negligible effects on soil bulk density, indicating that very low PES microfiber contamination (<0.3 %) did not have a considerable impact on bulk density in soil (Zhang et al., 2019). Recent studies have shown both positive and negative effects of microplastics on soil physical properties (Wang et al., 2021). For example, about 72% of plastic fiber particles were associated with soil aggregates and accumulated more in micro- than in macro-aggregates (Zhang and Liu, 2018). On the other hand, a reduction in the formation of large aggregates (>2 mm) and increased aggregate stability was observed with the presence of microfibers in soil. Microplastic films in soil facilitated artificial pore formation and prevented the formation of large aggregates, whereas microplastic foams promoted the formation of large aggregates and decreased aggregate stability (Lehmann et al., 2021). Size distributions and profiles of water-stable aggregates in soil have been shown to be altered by the presence of microplastics (Boots et al., 2019). The mean weight diameter of water-stable aggregates of soils without microplastics was greater than those of fiber-, HDPE-, and biodegradable polylactic acid (PLA)-treated soils. Soil without microplastics exhibited a greater number of macro-aggregates (>2000 μm) and fewer micro-aggregates ranging from 63 to 250 μm, compared to soils exposed to HDPE and PLA. The number of micro-aggregates (< 63 μm) decreased after each HDPE and PLA treatment. These results suggest that bonds between micro-aggregates are altered by soil microplastic contamination (Boots et al., 2019). Furthermore, aggregate size decreased with increasing concentrations of aggregate-associated plastic fiber, whereas lower concentrations of films and fragmented plastics were detected in micro-aggregates in arable land throughout southwestern China (Zhang and Liu, 2018).

Microplastic-enriched micro-aggregates may not tend to form macro-aggregates. However, in a pot experiment, water-stable macro-aggregates (>2 mm) increased as the concentration of PES microfibers increased, with the water stability of PES fiber-associated aggregates being enhanced over several drying and wetting cycles (Zhang et al., 2019). As a hydraulic effect of microplastics, evapotranspiration increased up to 35% with polyamide (PA) beads and 50% with PES, indicating that water availability was increased in the presence of microplastics. Hence, PES-treated soil demonstrated a significant increase in water-holding capacity, exhibiting higher water saturation over long periods (De Souza Machado et al., 2019). Another study showed that fine (<5 μm) PES fibers hindered pores smaller than 30 μm and may increase water repellency by influencing hydrophobicity (Zhang et al., 2019). Furthermore, the addition of PES fibers increased the number of macropores (>30 μm). Linear-shaped PES microfibers facilitated clumping by entangling soil particles, which may result in the formation of macropores (Zhang et al., 2019). Similarly, the water content of soil treated with film-type microplastics was lower than that of soil without microplastics (Wan et al., 2019).

Exposure to smaller microplastics accelerated the evaporation of water in soil. In soils with higher clay contents, evaporation rates were typically higher than those of sandy soils, likely because plastic particles tend to be larger than clay particles and therefore decrease soil water conductivity. Additional evaporation occurred when the proportion of microplastics in soil increased, leading to water shortages in soils. Moreover, microplastic contamination resulted in desiccation cracking in soils with high clay content, indicating the potential for the migration of pollutants and nutrients, and also increases in soil permeability. Evaporation likely results in microplastics altering the soil physical properties, including cohesion, moisture retention, and root penetration (Boots et al., 2019). Therefore, soil structure and water dynamics can be affected by certain microplastic types and forms. In contrast, microplastics of similar sizes, shapes, or chemical compositions exhibit weaker effects on soil structure and water dynamics. Because microplastics are associated with humic-like substances, their addition may improve soil water holding capacity, stability, and nutrient availability.

When considering the effects of microplastics on aggregate stability, physical interactions alone do not account for aggregate alterations and biological effects contributing to soil aggregation also need to be considered. Microplastics can affect microbial microhabitats and likely affect microbial contributions to soil aggregation (Lehmann et al., 2019). For example, hydrophobins, amphiphilic proteins secreted by fungi, are important for soil hydrophobicity and aggregate stability; their production may be restricted by microplastics (Rillig, 2005). Additionally, mispredictions of carbon concentrations in soil matrices can be ascribed to variations in soil bulk density, driven by the presence of microplastics (Rillig, 2018).

The above data demonstrate that alterations in soil function associated with microplastics are driven by their types, forms, and amounts. The changes they produce in soil physical properties can affect soil biota and other physical and biogeochemical processes in these ecosystems. However, only a few studies have investigated the effects of microplastics on soil physical properties; moreover, field studies have not been performed to evaluate environmentally relevant scenarios. Thus, further research is required to elucidate the effects of microplastics on the overall structure and function of soils.

Table 1. Effects of microplastics on soil physical properties

Type of polymer	Concentration	Soil texture	Duration	Type of study (Incubation/ field)	Effect	Ref.
Polyacrylic fibers, Polyethylene high-density fragments, Polyamide beads, Polyester fibers	0.05%, 0.10%, 0.20%, and 0.40% (soil dry weight basis) for polyamide and polyethylene 0.25%, 0. 50%, 1.00%, and 2.00% (soil dry weight basis) for polyester and polyacrylic	Loamy sand	5 weeks	Pot experiment	All particles affected soil bulk density Polyester fibers caused a concentration-dependent reduction in soil bulk density and water-stable aggregates, increasing water holding capacity.	(De Souza Machado et al., 2018)
Synthetic	HDPE, PLA 0.1 %,	Sandy	30 days	Pot experiment	pH of the soil when exposed to	(Boots et al.,

fibers, High-density polyethylene (HDPE), Polyacrylic (PLA)	Synthetic fibers 0.001 %	clay loam			HDPE was significantly lower than when it was exposed to all other treatments. Soils in control treatments had 24, 35, and 28% greater mean weight diameters than those in which fibers, HDPE, and PLA, respectively, had been added. Soil from controls had 60 and 53% more large macroaggregates (>2000 µm) than soils with HDPE and PLA, respectively. Conversely, micro-aggregates (250−63 µm) were significantly higher in soil exposed to all types of microplastics when compared to control soil.	2019)
Polyester microfiber	0, 0.1% and 0.3%	Clay	1 year	Field experiment	No detectable changes in soil bulk	(Zhang et al.,

		loam			density, soil aggregate size distribution, and saturated hydraulic conductivity in soils treated with different concentrations of polyester microfibers (PMFs). A significantly higher volume of >30 μm pores in the 0.3% PMF treatment (21.4%) was observed compared to those in the 0.1% PMF (11.6%) and non-polyester microfiber (10.3%) treatments after one year.	2019)
Polyester microfiber	0, 0.1% and 0.3%	Clay loam	75 days	Pot experiment	No differences in soil bulk density between the polyester microfiber (PMF) treatments and the non-polyester microfiber (NPMF) treatment during the incubation period. After 75 days of incubation,	(Zhang et al., 2019)

					significantly higher contents of large macroaggregates (>2 mm) were found in the PMF treatments compared to the NPMF (30.8%) treatment. In contrast, NPMF treatment significantly increased microaggregate (0.25–0.05 mm) and silt+clay (<0.05 mm) fractions than the PMF treatments.	
Polyester (PES) High-density polyethylene (HDPE), polypropylene (PP) polystyrene (PS), and polyethylene terephthalate (PET)	PES 0.2 % HDPE, PS, PP, PET 2 %	Loamy sand	2 months	Pot experiment	Soil bulk density was decreased by PEHD, PES, PET, PP, and PS. Significant decreases in water-stable aggregates were observed in soils with PA, PES, and PS. All microplastic treatments altered soil structure, with the intensity of effects depending on the microplastic	(De Souza Machado et al., 2019)

	type.

3.2 *Effects on soil chemical properties*

Although plastic is generally resistant to degradation in soils, significantly degraded-microplastic particles are labile and have important chemical interactions with soil particles. Once microplastics break down into smaller sizes, chemical reactions occur through several unpredictable and poorly understood mechanisms based on polymer composition and soil type. For example, after exposure to different types of microplastics for one month, the pH of HDPE-treated soils (6.35±0.14) was slightly lower than that of controls (6.96±0.02), whereas no significant difference was observed with PLA (Boots et al., 2019). In another study, contamination of soil with microplastics derived from low-density polyethylene (LDPE) and bio-plastic (1%, w/w) increased soil pH (Qi et al., 2020). Palansooriya et al. (2022a) observed a significant reduction in pH in soil contaminated with LDPE ≥1% (w/w). On the other hand, exposure to 1% (w/w) LDPE has not significantly altered soil pH in agricultural soil (Dissanayake et al., 2022; Palansooriya, 2022b). Exposure to polyethylene microplastics reduced the pH of an acidic soil, but increased pH in an alkaline soil, indicating that the effect of microplastics on soil pH could vary depending on soil types (H.-Z. Li et al., 2021). Changes in soil pH have been shown to affect soil nutrient availability, consequently affecting plant growth and crop productivity (Wang et al., 2021). On the other hand, residual monomers of PA particle surfaces have been shown to leach nitrogen into soil, acting as a fertilizer (De Souza Machado et al., 2019). Similarly, primary polymer-based pellets can release organic phosphite antioxidants that degrade into organic phosphate in the soil. Carbon in microplastics, which is typically inert, may influence the microbial availability of carbon in soils and have critical consequences for biogeochemical carbon cycling.

The decomposition of plastics (e.g., LDPE) is a time-dependent process, and likely drives microbial immobilization of nutrients (Rillig et al., 2019). Conversely, carbon in HDPE would be less likely to undergo biogeochemical alterations in soils. Therefore, microplastics could act as soil carbon reservoirs, complicating soil carbon storage estimates (Rillig, 2018). Carbon in microplastics might be confused with carbon stored in the soil by more natural mechanisms; accordingly, microplastic-driven carbon should be excluded from such estimates to enable policymakers to predict soil carbon retention properties more realistically.

Dissolved organic matter (DOM) is a critical driver of soil biogeochemical processes, often mediating overall organic carbon cycles as well as nutrient migration. With higher microplastic addition, increased nutrient content of DOM was observed in the form of dissolved organic carbon, dissolved organic nitrogen, dissolved organic phosphorus, nitrate, phosphate, high molecular weight humic-like substances, and fulvic acid (Liu et al., 2017). Soil organic-matter content, which was assumed to contain plastic mulch residues, was lower than that in bare soils without mulch residues (Hou et al., 2019). Previous studies did not observe significant changes in available soil nutrients due to microplastic exposure at environmentally relevant concentrations (H.-Z. Li et al., 2021). For example, the addition of 0.2% polyethylene microplastics to soil did not significantly affect dissolved organic carbon, $NH^{+}{}_{4}$-N, and Olsen P concentrations (H.-Z. Li et al., 2021). On the contrary, a significant increase in NO_2^{-}-N content but a significant decrease in NH_4^{+}-N content was observed with soil exposure to PLA microplastics (2% w/w) (Chen et al., 2020). The high proportions of microplastics (28% w/w) significantly increased dissolved organic carbon, dissolved organic nitrogen, dissolved organic phosphorus, and PO_4^{3-} concentrations (Liu et al., 2017). In addition, microplastics have been shown to alter soil microbial communities, subsequently influencing soil nutrient cycling (Ren et al., 2021).

Nonetheless, data to develop a comprehensive consensus regarding the overall and complete effects of microplastics on soil chemical properties remain insufficient. For example, although it is well-established that microplastics alter soil pH, the mechanisms underlying these changes remain poorly understood. Hence, both field and laboratory studies on a broad range of soil and microplastic types are needed, using environmentally relevant concentrations of microplastics to obtain a clearer picture of their effects on soil properties. Moreover, most of the current available research data are sourced from laboratory experiments; therefore, more field (*in situ*) experiments are needed to further assess the microplastic effects on soils. However, challenges remain regarding the lack of robust methods for sampling, classification, and analysis of microplastics in soil (Gong and Xie, 2020). Several analytical methods to assess microplastics in soil have recently been developed, but an absence of standardized methods (Provencher et al., 2020) hinders the accurate assessment of microplastic effects on soil ecosystems; hence, comparison between studies is difficult.

3.3 ***Effects on soil microbial properties***

Microorganisms in soil are important in maintaining soil structure, organic matter decomposition, and nutrient cycling. The spatial distributions of soil microbial (bacterial and fungal) communities and microbial activities are closely related to soil properties such as organic matter content, texture, and moisture content in agricultural environments (Naveed et al., 2016; Rillig et al., 2017b).

Changes in soil habitat are considered to be one of the primary drivers of soil microbial activity (Zhou et al., 2020). Soil porosity and moisture can be altered by microplastics via changing oxygen concentrations. These changes affect the relative abundance of both aerobic and anaerobic microorganisms (Rubol et al., 2013). De Souza Machado et al. (2018) demonstrated that multiple microplastic types (PES, polyacrylic, PE, and PA) affected soil physical environments and increased microbial activity, thereby improving soil aggregate structure. Indigenous microorganisms may lose their microhabitats due to changes in pore spaces by microplastics, leading to reduced abundance and biomass (Veresoglou et al., 2015). Bandopadhyay et al. (2018) suggested that biodegradable plastic mulches altered soil microclimates and physical structures, affecting the composition of soil microbial communities by creating new ecological niches. Moreover, soil pH may be changed by the presence of microplastics, resulting in the alteration of microbial communities. However, the mechanisms and drivers of soil pH changes caused by microplastic addition are still not well understood (Khalid et al., 2020). Furthermore, changes in soil enzymatic activities, including those of urease, fluorescein diacetate hydrolase, phenol oxidase, and catalase, have been observed in the presence of microplastics (Huang et al., 2019; Liu et al., 2020). Soil microbial function and community structure can also be affected by microplastic-driven decreases in respiration rates (Judy et al., 2019).

Several laboratory studies have demonstrated that microplastics can exhibit toxic effects on yeasts and fungi according to surface charge properties, particle size, salt concentration, and microorganism species composition (Miyazaki et al., 2014; 2015; Nomura et al., 2016). These results imply that microplastics may change the distribution and abundance of yeast and fungi in the field. De Souza Machado et al. (2019) reported that root colonization by arbuscular mycorrhizal fungi varied according to microplastic type. Conversely, the experimental addition of microplastics to a mixture of organic waste and soil led to a decrease in archaea and an increase in fungal populations, demonstrating the potential effects of microplastics on microbial community structure (Judy et al., 2019). However, this investigation did not report any effects of microplastics on the abundances of bacterial 16S rRNA sequences, nitrogen cycle-related functional genes, and substrate-induced respiration.

Dissolved organic matter in soil derived from microplastics acts as a source of carbon and also serves as a substrate for soil microorganisms; therefore, changes in DOM may affect microbial activity and substrate availability (De Souza Machado et al., 2019). Rillig et al. (2018) reported that microplastic-derived carbon may be perceived as a soil organic carbon source because of its high carbon content. As soil microorganisms involve in primary decomposition of organic matter, they may detoxify harmful chemicals including microplastics, which can be viewed as an ecological “plastisphere” for soil microorganisms (Oberbeckmann et al., 2016). Plastispheres often exhibit low microbial abundance and high microbial community homogeneity (Zettler et al., 2013). *Vibrionaceae* and *Pseudoalteromonadaceae* were the most common bacterial taxa on plastic debris that were not detected in the surrounding environment. Correspondingly, soil microorganisms may play important roles in the transformation of both macroplastics and microplastics (Helmberger et al., 2020), particularly by contributing to microplastic degradation (Yuan et al., 2020). For example, *Rhodococcus ruber* C208, isolated from mulch film-buried soils, readily colonized PE surfaces and degraded photooxidized PE (Gilan et al., 2004), demonstrating the enzymatic and genetic basis (Gravouil et al., 2017; Santo et al., 2013). *Brevibacillus borstelensis* has been reported to degrade PE (Hadad et al., 2005), and *Comamonas acidovorans* TB-35 was shown to use PES polyurethane as a source of carbon by producing a polyurethane-degrading enzyme (Akutsu et al., 1998). During degradation, enzymes secreted by microorganisms likely facilitate microbial attachment to microplastic surfaces and biodegradation of polymers into hydrophilic, micromolecular intermediates. As a result, the intermediates are taken up by cells and metabolized, releasing CO_2, H_2O, and CH_4 as the end products (Nizzetto et al., 2016).

Soil microorganisms may be negatively affected by specific selective pressures caused by the occurrence and abundance of microplastics, potentially changing diversity, community structure, and evolution (Rillig, 2018). Nevertheless, the impact of microplastics on soil microbial properties are still not well understood. Considerable uncertainty regarding microplastic effects on soil microorganisms still exist due to a lack of information on the interactions between microplastics and soil organisms. Many scientists have suggested that microplastics may pose a serious threat to soil organisms by reducing their growth and reproduction, thereby reducing biodiversity in soils. In contrast, others have reported negligible effects of microplastics on soil organisms (Lwanga et al., 2016; Rodriguez-Seijo et al., 2017; Wang et al., 2019).

To improve risk assessments of the impacts of microplastics on soil organisms and subsequent effects on ecological processes, further research is required to address individual factors, such as microplastic type, form, and particle size, plant genetic variation, and soil conditions, and their relationships to soil factors, to systematically evaluate the full suite of potential consequences in terrestrial environments. Weathered microplastics should be used in future experimental studies, since weathering changes particle size and surface charge, which likely affect their uptake and metabolism. Weathering produces smaller particles that are likely more bioavailable and easily assimilated by organisms, while altered surface charges can affect microplastic aggregation and dispersion dynamics in organisms (Fu et al., 2019) and changes in cell toxicity (Kim et al., 2017). The release of chemicals such as additives and microplastic-derived intermediates by weathering should also be considered to better evaluate their direct and indirect risks (Liu et al., 2020). Overall, further research on interactions between microplastics and soil biota will improve our understanding of impacts in both naturally and anthropogenically influenced environments.

4. Interactions between microplastics and other contaminants

4.1 *Interactions with organic contaminants*

Microplastics can act as both sources and sinks of organic contaminants. Multiple toxic organic contaminants can be present in plastics that are released by weathering and degradation in both terrestrial and aquatic environments. Conversely, microplastics can adsorb and concentrate organic contaminants on their surfaces while dispersed in soil and aqueous media, serving as a sink. These bound contaminants can then be transported within and across ecosystems (Bank and Hansson, 2019), potentially increasing offsite environmental risks.

Several chemicals are intentionally added to plastics during manufacturing to improve their performance, flexibility, stability, and functionality (Hahladakis et al., 2018). Hazardous substances including plasticizers, flame retardants, surfactants, solvents, stabilizers, colorants, and biocides are used as enhancements (Groh et al., 2019; Mandal et al., 2020). Many of these chemicals are extremely persistent in the environment and harmful to organisms and food webs. Well-known toxic organic chemicals found in plastic products include PAHs, pesticides, and PCBs, all of which can be released into soil and aquatic ecosystems. As they are present in multiple plastic products, phthalate-based plasticizers (e.g., ethylhexyl phthalate, dimethyl phthalate, diethyl phthalate, di-n-butyl phthalate, benzyl butyl phthalate, bis-2-ethylhexyl adipate, and di-n-octyl phthalate) are often found in microplastic-contaminated soil, sediment, water, and sludge. For example, Borges Ramirez et al. (2019) reported a positive correlation between the microplastic abundance and phthalate ester compounds in drainage systems and coastal sediments in Mexico (Table 2). Similarly, tris-(2-chloroethyl)-phosphate, tris (1-chloro-2-propyl) phosphate, and di-(2-ethylhexyl) phthalate were the most dominant plasticizers found in microplastics collected from the beaches of the Bohai Sea and the Yellow Sea in northern China (Zhang et al., 2018). Brominated flame retardants (e.g., polybrominated diphenyl ethers (PBDEs) and 1,2-bis(2,4,6-tribromophenoxy) ethane (BTBPE)) are another group of toxic additives used during plastic manufacturing to reduce the flammability of the final product. These chemicals do not form any bonds with the original polymer and thus can be released over time under altered chemical conditions in the environment (Sun et al., 2019). Their release from their original plastic matrix can be slow, but the rate can increase significantly upon the destruction of the structural integrity of the matrix due to fragmentation, pulverization, wear and tear, and environmental weathering (Gaylor et al., 2013). Landfills and biosolids, which are known microplastic hotspots, are critical non-point sources of plastic-derived persistent chemical contamination by substances including brominated flame retardants (McGrath et al., 2017) and per- and polyfluoroalkyls (Hepburn et al., 2019).

When microplastics act as sinks, their physicochemical characteristics govern the fates, transformation, transport, and bioavailability of organic contaminants. Specifically, the hydrophobicity of microplastic surfaces plays a key role in the adsorption behavior of contaminants, because additive and plasticizer chemicals added in plastics are predominately hydrophobic (Kwon et al., 2017). This adsorption of organic contaminants is empirically expressed as the partition or adsorption coefficient at equilibrium, where the diffusion of chemicals within the microplastic interior is often the rate-limiting factor. This diffusion is equally important during the transfer of organic contaminants from water to microplastics and vice versa (Kwon et al., 2017).

Numerous studies have investigated the adsorption of organic contaminants by microplastics. Wang et al. (2015) reported strong adsorption of phenanthrene and nitrobenzene by PS micro(nano)plastics of various sizes (50 nm to 170 μm) with log K_d values of 3.07–4.20 and 1.58–3.14, respectively. Hydrophobic partitioning was the primary mechanism driving phenanthrene and nitrobenzene adsorption, and smaller plastic particles had higher log K_d values. However, a deviation in the effect of particle size on adsorption of organic contaminants was observed due to the rapid aggregation of nanoplastic particles, reducing the effective surface area available for contaminant adsorption (Wang et al., 2019). Microplastic chemistry, particularly polymer structure and molecular composition, also affects the adsorption of persistent organic pollutants. The adsorption of perfluorooctanesulfonamide on microplastics followed the sequence of PE > polyvinyl chloride (PVC) > PS, according to their chemical compositions (Wang et al., 2015). The most prevalent adsorption property was primarily hydrophobic, with greater K_d values for perfluorooctanesulfonamide than for perfluorooctanesulfonate for all three microplastic types. Because perfluorooctanesulfonate contains an anionic sulfonate, its adsorption was influenced by solution pH and its overall ionic strength. In contrast, these effects were negligible for perfluorooctanesulfonamide, where only the hydrophobic interaction governed adsorption. The adsorption of perfluorooctanesulfonate increased at lower pH and higher electrolyte concentration because of the partial involvement of electrostatic attraction and hydrophobic interaction (Wang et al., 2015).

Microplastics can also adsorb hydrophilic organic contaminants, which can be attributed to the surface formation of oxygen-containing functional groups. For example, Liu et al. (2019)

reported that aging of microplastics by UV light significantly oxidized PS and PVC surfaces, creating minuscule localized cracks that subsequently enhanced the adsorption of the polar contaminant ciprofloxacin, compared to that of the pristine microplastics. Electrostatic interactions, intermolecular hydrogen bonding, and partitioning were identified as the dominant hydrophilic mechanisms for ciprofloxacin adsorption. Because electrostatic attraction was involved in adsorption, pH changes in the solution changed the extent of ciprofloxacin adsorption because the adsorbate molecular charge changed (i.e., pK_a value) as a function of pH (Liu et al., 2019).

Adsorbed contaminants can be desorbed from microplastics under varying environmental conditions, including changes in pH, temperature, and ionic strength. Microplastics coated with organic pollutants can be ingested by aquatic organisms (Duis and Coors, 2016). The chemical gut environment after ingestion can also enhance release of organic contaminants, thereby increasing their bio-accessibility. For example, Bakir et al. (2014) reported that in the presence of a gut surfactant under simulated gut conditions, increased temperature enhanced the release of adsorbed DDT, phenanthrene, perfluorooctanoic acid, and di-2-ethylhexyl phthalate from PVC and PE microplastics, indicating a greater bio-accessibility of these plastic-derived contaminants for warm-blooded than for cold-blooded organisms. The presence of gut surfactant resulted in a nearly 30-fold increase in the desorption of organic contaminants over that in normal seawater conditions (Bakir et al., 2014).

In summary, microplastics can serve both as sources and sinks of organic contaminants in the environment. The release of parent organic chemicals from microplastics is diffusion-limited, with weathering and structural damage enhancing release rates. The strong affinity of microplastics for organic contaminants simultaneously enables them to concentrate these contaminants on their surfaces and transport them offsite, acting as a “Trojan horse.” The physicochemical characteristics of particles and solution chemistry govern the binding of contaminants, greatly influencing their bioavailability and bio-accessibility to target organisms.

Table 2. Interactions of microplastics with organic pollutants

Pollutant Type	Pollutant concentration	Microplastic category	Microplastic concentration	Microplastic size	Microplastic shape	Effect	References
Phthalate esters	1.96–21.7 μg/g	Not specified	76–472 items/m^2	0.3–5 mm	Fibers, chips, and fragments	Di-2-ethylhexyl phthalate and di-n-octyl phthalate were most frequently detected. A positive correlation was observed between microplastic abundance and phthalate ester concentrations in marine coastal and urban channel areas in Mexico.	Borges Ramirez et al., 2019
Organophos phorus	OPE: 0–84.6 μg/g	Polystyrene, polyethylene,	5–1090 items/kg of	<1–5 mm	Pellets, fragments,	Tris-(2-chloroethyl)-phosphate, tris (1-chloro-	Zhang et al., 2018

esters (OPEs) and phthalic acid esters (PAEs)	PAE: 0–80.4 ng/g	and polypropylene	sediment	flakes, and foams	2-propyl) phosphate and di-(2-ethylhexyl) phthalate were the most abundant compounds found.
Polybrominated diphenyl ethers (PBDEs) and 1,2-bis(2,4,6-tribromophenoxy)ethane (BTBPE)		Acrylonitrile butadiene styrene			Leaching rates of brominated flame retardants from microplastic pellets made of acrylonitrile butadiene styrene were controlled by their diffusion within the plastic matrix.

4.2 *Interactions with inorganic contaminants*

The interactive and cumulative effects of different microplastic types are emerging themes within this sphere of environmental research (Wang et al., 2019b). Due to their larger surface areas and hydrophobicity, microplastics can absorb harmful chemicals from soil solutions to concentrate them locally (Rillig, 2012). Microplastics can interact with metals through direct physical adsorption, binding to charged sites or neutral regions, and coprecipitation and adsorption onto hydrous oxides (Bradney et al., 2019). If chemicals retained by microplastics are transferred through trophic levels, microplastics can act as a vector for these chemicals in terrestrial environments, making them bioavailable to soil organisms. However, microplastics play a negligible role as a vector compared to natural granular materials due to their low abundance in terrestrial environments, leading to dilution, competitive sorption, and other mitigations (Hartmann et al., 2017; Hüffer et al., 2019). Therefore, there is considerable uncertainty regarding the extent to which microplastics influence pollutant effects.

As microplastics undergo degradation when exposed to UV radiation and microorganisms under oxidative conditions, changes in their surface area and molecular polarity may affect their adsorption of contaminants (X. Wang et al., 2021). The polymeric chains of microplastics are broken with aging, resulting in increased surface roughness, oxygen-containing functional groups, and surface area (Ren et al., 2021). Thus, microplastic aging may pose a substantial threat to the terrestrial ecosystems via their increased adsorption capacity for multiple pollutants, including heavy metals and metalloids.

Contrarily, DOM enrichment triggered by microplastic addition can contribute to decreased soil adsorption of contaminants. Therefore, the accumulation of DOM mobilizes several soil contaminants, affecting their toxicity and bioavailability (Liu et al., 2017). Fulvic acid-like materials derived from DOM can facilitate the migration and bioavailability of heavy metals by serving as their carriers.

Heavy-metal contamination is prevalent across urban and agricultural soils; however, only a few studies have addressed the effects of metal sorption on microplastics. When considering the simultaneous presence of both heavy metals and microplastics as contaminants, their geochemical interactions may lead to unpredictable behaviors in soil environments. This process also poses an emerging threat since heavy metals bound to microplastics can be released into the surrounding environment.

When both soil and microplastics (HDPE) co-occur in a mixture, the level of Zn on the microplastics was lower than that absorbed by soil (Hodson et al., 2017). Although Zn binds to microplastics, a significant amount is readily desorbed compared to that bound in soil. Correspondingly, the addition of HDPE minimized the adsorption of Cd in soils (dilution effect), which can be attributed to its hydrophobic surface with simpler characteristics than those of soil surfaces (Zhang et al., 2020a). In addition, the effect of microplastics on heavy-metal absorption by soil varies substantially and depends on size, dose, time of exposure, and soil pH. Cd absorption by soil decreased with increasing microplastic size and dose; moreover, an increase in pH led to significant increases in Cd absorption by soil, suggesting more complex surface properties. Hence, a reduction in pH driven by microplastic addition may elevate the labile form of certain heavy metals. The highly crystalline features of microplastics can account for their lower absorption capacities relative to those of soils, although the desorption of Cd from a soil–

microplastic mixture is correlated with the size and dose of the microplastics. Moreover, the aging induced by ultraviolet radiation likely leads to surface modification that further alters the absorptive capacities of soils and microplastics.

However, it has been reported that metals added during microplastic manufacturing are the primary source of heavy metals transported by microplastics (Wang et al., 2017). In addition, alkaline components of microplastics play a vital role in heavy metal adsorption. Heavy-metal adsorption can vary with microplastic physical and chemical properties. Hence, plastic type is less important in heavy-metal adsorption than physicochemical properties (Bradney et al., 2019). Field monitoring data revealed that microplastic heavy-metal content was proportional to heavy-metal soil pollution (Zhou et al., 2019). Furthermore, metal content of microplastics was correlated with soil particle number. Consequently, heavy metals absorbed by microplastics can have adverse effects on soil organisms, which can further facilitate the availability of heavy metals throughout the food web. Hence, further studies are needed to identify the long-term stability and bioavailability of heavy metals that are transported by microplastics.

5. Ecotoxicological impacts of microplastics on terrestrial plants and other organisms

5.1 *Impacts on terrestrial plants*

The potential impacts of microplastics on terrestrial plants are not well understood, and related research findings are currently insufficient (Wang et al., 2020). In general, microplastics in soil can induce changes in properties such as moisture, density, structure, and nutrient content, which may in turn alter plant root characteristics, growth, and nutrient uptake (De Souza Machado et al., 2019; Qi et al., 2018; Rillig et al., 2019). Studies have demonstrated that microplastics impact wheat (*Triticum aestivum*) (Qi et al., 2018), spring onion (*Allium fistulosum*) (De Souza Machado et al., 2019), cress (*Lepidium sativum*) (Bosker et al., 2019), and faba bean (*Vicia faba*) (Jiang et al., 2019). These results suggest that plant responses are dependent on species, soil, and microplastic properties.

The accumulation of microplastics in plants may hinder cell–cell contacts or block cell-wall pores, restricting the transport and absorption of essential nutrients (Asli and Neumann, 2009; Ma et al., 2010). PS microplastics damaged *Vicia faba*, i.e., growth retardation and genotoxic impairment, when a large amount of them (~100 nm) accumulated in roots (Jiang et al., 2019). A recent study on *Lepidium sativum* (cress) revealed that the accumulation of microplastics on seed capsules (~4.8 μm) significantly decreased the germination rate after 8 h of exposure (Bosker et al., 2019). A significant difference in root growth was also reported in the same study upon microplastic exposure for 24 h. The effects of microplastics on vascular plants remain ambiguous, and research with clear evidence remains limited. A few studies have revealed considerable impacts of microplastics on plants such as *Allium fistulosum* (spring onion) (de Souza Machado et al., 2018), *Lolium perenne* (perennial ryegrass) (Boots et al., 2019), and *Triticum aestivum* (wheat) (Qi et al., 2018). De Souza Machado et al. (2019) observed that microplastics (PES fibers, polyamide beads, PE, PES terephthalate, PP, and PS) altered the characteristics of spring onion (*Allium fistulosum*), including root and leaf traits; total biomass; and leaf composition, such as nitrogen content and the carbon-to-nitrogen (C:N) ratio. They proposed a casual model for the effects of microplastics in terrestrial ecosystems; microplastics occur in a cascade of shifts in the biophysical environment of soil, which affects onion growth. In contrast, Judy et al. (2019) observed that the addition of microplastics (HDPE, PET, and PVC) produced no significant negative effects on seedling emergence and wheat biomass production.

These results indicate that additional research is required to assess the microplastic impacts on plants and their relationships with soil ecosystems. In another study, the type of plastic mulch film was found to significantly affect wheat growth; compared to those derived from LDPE, microplastics from starch-based plastic mulch films (37.1% pullulan, 44.6% PET, and 18.3% polybutylene terephthalate) exhibited strong negative effects on wheat during both vegetative and reproductive growth (Qi et al., 2018). Only a few studies have reported the impacts of microplastics on plants and in vivo transport (Zhou et al., 2020). The interactions among microplastic characteristics (type, concentration, and source), soils, and plants remain largely unknown. Therefore, further research is needed to understand the response mechanisms of various crops to address the knowledge gaps pertaining to the impacts of microplastics on plants. Moreover, microplastic accumulation in plants can pose risks to terrestrial organisms through trophic transfer (Kumar et al., 2020). Therefore, it is important to focus future research efforts on how microplastic pollution affects local food webs.

5.2 *Ecotoxicological impacts on invertebrates and vertebrates*

Soil organisms, such as earthworms, mites, and collembola, are essential for the maintenance of soil quality and proper ecosystem functioning. However, microplastics can pose a major threat to these organisms (Qi et al., 2020). Multiple factors, including type, size, form, shape, color, density, abundance, physical behavior, chemical composition, aging, and biological interactions, affect microplastic ecotoxicity (Wright et al., 2013). For instance, it has been hypothesized that smaller microplastics are more toxic than larger ones, because the former may cause interior abrasions and obstructions in organisms. Smaller microplastics may pass through the protective cell wall and cell membrane, resulting in harmful organismal effects (Wu et al., 2021).

Moreover, microplastic ecotoxicity varies by plastic type depending on their chemical composition (Zimmermann et al., 2020). As most of the additives included in plastics are bound through weak van der Waals forces, they can leach into the environment during aging, posing toxic effects to the organisms (Zimmermann et al., 2020). In many cases, soil biota uptake microplastics as food, inducing a decrease in carbon biomass ingestion that can lead to energy depletion, inhibited development, and even mortality (da Costa et al., 2016).

Soil animals may play a vital role in microplastic movement and distribution. Earthworms can transport microplastics in soils through adhesion and excretion, causing smaller microbeads to migrate largely downward in the soil profile (Rillig et al., 2017). Rodriguez-Seijo et al. (2017) observed that there was no significant effect on the survival, reproduction, and growth of earthworms in soil exposed to 0–1000 mg/kg of polyethylene pellets after 28 days of exposure. In this study, a maximum 1000 mg/kg microplastic loading was tested due to its ecological relevance (Rodriguez-Seijo et al., 2017). However, they observed gut damage in earthworms after exposures to as little as 125 mg/kg of microplastics. Histopathological damage was observed at a concentration as low as 62.5 mg/kg, which could be more severe at higher concentrations (Rodriguez-Seijo et al., 2017). Wang et al. (2019) found that the oxidative stress in earthworms (*Eisenia fetida*) caused by microplastics had no clear pattern; they suggested that soil microplastics may have negligible effects on earthworm fitness and hydrophobic contaminant bioaccumulation. Trestrail et al. (2020) found that microplastic ingestion by invertebrates disrupted redox homeostasis, as the ingested microplastics can cause oxidative stress and trigger antioxidant upregulation in these organisms. Moreover, *C. elegans* exposed to nano- and micro-PS displayed size-dependent neurotoxicity in cholinergic and GABAergic neurons, as well as oxidative damage (Lei et al., 2018a). Furthermore, earthworms exposed to soil with 2% PS microplastic exhibited a mortality rate of approximately 40% (Cao et al., 2017). Nevertheless, Zhou et al (2020), reported that 2% microplastic contamination is extremely high and unlikely to be environmentally relevant.

Microarthropods can transport and distribute microplastics in soil and improve certain predator–prey relationships (Zhu et al., 2017). For example, mites can scrap, chew, and distribute microplastics (Maaß et al., 2017). Zhu et al. (2018) observed that oribatid mites transport microplastics up to 9 cm, changing soil moisture and structure. Moreover, abnormal lipid metabolism in mice was observed at the cellular level through physiological and biochemical indicators. The modulation of mouse hepatic triglyceride and total cholesterol upon exposure to multiple sizes of microplastics was investigated (Deng et al., 2017). Oxidative stress was found to be the key toxic effect; after exposure, hepatic oxidative stress was detected using superoxide dismutase, catalase, and glutathione peroxidase biomarkers (Deng et al., 2017). Metabolites related to oxidative stress were detected in the serum of the mice. Furthermore, the presence of microplastics may aggravate the toxicity of other environmental pollutants (Browne et al., 2013; Deng et al., 2018), and microplastics adhering to the organisms' exterior surfaces may hamper their movement (Kim and An, 2019).

Although studies supporting the potential health risks and accumulation of microplastics in terrestrial animal tissues are scarce, these risks should not be overlooked, particularly given the increasing stockpile of environmental plastic waste (Jambeck et al., 2015). Microplastics can transfer from lower (prey) to higher (predators) trophic levels in food webs (Guo et al., 2020). When the time consumed from ingestion to egestion is less than the retention time of microplastics in the organs of prey, microplastics are ingested and concentrated by predators (Guo et al., 2020). Consequently, biomagnification can occur in natural food webs in terrestrial ecosystems (Guo et al., 2020). Microplastics ingested by micro- and mesofauna resulted in food-chain contamination (Rillig, 2012), indicating potential adverse health effects on humans and other organisms (Li et al., 2020). Microplastic accumulation in tissues may induce multiple adverse effects, including neurological damage, oxidative stress, and changes in membrane permeability, energy metabolism, antioxidative capacity, and histology (de Souza Machado et al., 2018).

At present, there are a limited number of studies on the effects of microplastics on humans. Microplastics can enter into the human body via inhalation of PM 2.5, dermal exposure, and direct ingestion through drinking water or diet (Zhou et al., 2020). Table salt, tea bags, sea food, milk, and honey have been identified as major sources of microplastics that contaminate human diets (Zhang et al., 2020b). In a study conducted to assess the abundance of microplastics in 39 different brands of table salt in 16 countries of the world, microplastic concentrations varied between 0 and 1674 particles/kg; salts produced in Asian countries had higher concentrations (Kim et al., 2018). Moreover, it has been estimated that approximately 2.3 million micro-sized particles are released when one cup of tea is prepared using one tea bag (Hernandez et al., 2019). Lwanga et al. (2017) observed microplastic concentrations of 0.87 ± 1.9 particles/g in soil, 14.8 ± 28.8 particles/g in earthworm casts, and 129.8 ± 82.3 particles/g in chicken feces, indicating the trophic transfer of microplastics through the food chain.

Gaylor et al. (2013) demonstrated that earthworms ingested biosolids or polyurethane foam, leading to the accumulation of PBDE in their bodies. PBDE is used as a flame retardant and is toxic to humans as well. This study demonstrated that microplastic additives may be released in the environment and affect terrestrial organisms and humans. However, the release of toxic chemicals from microplastics in terrestrial ecosystems, as well as their combined and all-time toxicities, are not well understood. These knowledge gaps and challenges regarding the health risks of microplastics must be addressed in future research. Therefore, it is essential to assess the interactive and synergistic toxic effects of microplastics in conjunction with those of other pollutants on terrestrial biota, including humans.

6. Management of microplastics in terrestrial environments

Considering the risks associated with the microplastic pollution in soils, it is essential to focus on management and remediation of microplastic contaminated environments. Even though research on ecological and health risk associated with microplastics pollution is being widely performed throughout the world, there is a lack of focus on mitigation and remediation strategies (Zhou et al., 2020).

Landfilling has been employed as the end of plastic wastes. However, this requires a large space which ultimately result in severe environmental impacts and effects through contamination of the surrounded area (Narancic et al., 2020). Development of proper plastic waste management practices, reduction of plastic usage, and reuse and recycling of plastics are considered to be the best possible strategies for the mitigation of soil microplastic pollution (Bhatt et al., 2021; Yuan et al., 2021b). Mechanical recycling has been identified as the most economical method for recycling plastics (Gopinath et al., 2020); it includes collection, washing, sorting according to plastic type, grinding into smaller fragments, and remolding (Narancic et al., 2020). However, separation of plastic waste from other solid wastes is plagued by significant challenges (Armenise et al., 2021). In addition, tertiary recycling technologies, such as depolymerization, plasma-arc gasification, and pyrolysis have garnered interest for use at commercial scales (Armenise et al., 2021). Nevertheless, it is difficult to convert the whole mass of plastic into reusable products in the recycling process; moreover, the durability of recycled products tends to be lower than that of the original products (Gopinath et al., 2020).

Advanced and efficient catalytic conversion methods for upcycling of plastic waste may increase profitability and environmental sustainability (Li et al., 2021). Hence, governments should implement and promote these upcycling technologies in waste-management programs (Yuan et al., 2021b). Mixing of waste plastics with concrete has been demonstrated to improve concrete properties (Gopinath et al., 2020; Hama and Hilal, 2017).

Pyrolysis of plastic waste, which degrades high-molecular-weight plastic polymers into lighter liquid hydrocarbons (fuels), gaseous hydrocarbons, and char under oxygen depleted conditions, is another promising approach to minimize plastic pollution (Miandad et al., 2016; Wang et al., 2021). This could be achieved through thermal pyrolysis, in which high temperature and pressure are applied to plastic waste, or catalytic pyrolysis, in which a catalyst is used to increase degradation efficiency and reduce energy requirements (Gopinath et al., 2020). Copyrolysis of plastic waste through blending with other organic biomass, such as crop residue and animal manure, has been identified as a viable method for waste valorization (Rathnayake et al., 2021; Rentizelas et al., 2018; Sanchez-Hernandez et al., 2021; Singh et al., 2021). Biochar has been identified as a potential material for the immobilization of organic contaminants in soil and water (L. Wang et al., 2021; Yuan et al., 2021a). Owing to the organic nature of microplastics, biochar could be useful in the removal of microplastics from soil and water (Wang et al., 2021).

At present, there is extensive interest in bioplastics as an environmentally friendly solution and alternative for traditional plastics (Narancic et al., 2018). Even though bioplastics are produced from biomass, their biodegradability depends on the chemical structure of the polymer (Bhatt et al., 2021).

Life-cycle assessment (LCA) is an effective tool for evaluating the environmental impacts of materials or processes (Davidson et al., 2021; Ye et al., 2017; Yuan et al., 2021c). LCA of plastic waste management technologies can be used to compare waste-management strategies to identify environmentally sound solutions. Adopting the best strategies identified through LCA could reduce the risks associated with microplastics in the environment. Consequently, this will help to achieve the United Nations Sustainable Development Goals (UN SDGs), including SDG 3, "Good health and wellbeing;" SDG 6, "Clean water and sanitation;" SDG 15, "Responsible consumption and production;" SDG 14, "Life below water;" and SDG 15, "Life on land" (Sarkar et al., 2021).

The initiation of more stringent policies, coupled with the combined action of all stakeholders, is essential to reduce improper disposal of plastic waste and ensure a sustainable future (Kumar et al., 2020). Although the mitigation of plastic pollution is urgently needed, there remains a lack of the knowledge necessary for implementing mitigation decisions. Therefore, it is essential to initiate global monitoring of plastic pollution that considers the plastic waste trade concomitantly with environmental biomonitoring (Bank et al., 2021).

7. Conclusions

Microplastics can enter terrestrial ecosystems through a multitude of pathways and from a wide array of sources. They can accumulate in soil, producing both beneficial and detrimental effects on its physical, chemical, and biological properties. Furthermore, microplastics interact with multiple persistent organic pollutants, heavy metals, antibiotics, and other toxic chemicals, potentially harming soil ecosystem health and biota.

Currently, there are no standardized protocols for the extraction and quantification of soil microplastics. Hence, it is essential to develop precise, feasible, and reproducible methods for their extraction and quantification in soil ecosystems considering a wide array of particle sizes, polymer types, and forms. Moreover, the microplastic characteristics in terrestrial environments, their ecological effects, and their underlying mechanisms have not been adequately explored. Further studies are essential for understanding the effect of microplastic soil contamination on plant performance and crop productivity in agricultural environments. Since microorganisms play an important role in the degradation of microplastics in soil, it is essential to understand their roles in degradation. After microplastics enter the terrestrial environment, their trophic transfer (along with that of their associated contaminants) through the food web can potentially affect both human and ecosystem health. Thus, it is important to examine the bioavailability and bioaccumulation of microplastics in terrestrial biota, while considering the potential for both direct and indirect effects. At present, the scientific community mainly raises concerns about the identification, quantification, and impact assessment of plastics in terrestrial ecosystems. However, these issues are increasing because of continuous, worldwide mismanagement of plastic wastes. Therefore, technological breakthroughs are urgently needed to foster environmentally friendly and economically sound plastic waste management strategies.

Scientists, policymakers, and the public must collaborate to help mitigate the accumulation of microplastics in terrestrial environments by implementing international policies and legislation designed to minimize microplastic pollution. The development of proper waste management strategies can reduce the risks associated with microplastic contamination in terrestrial ecosystems. Hence, all governments need to ensure the implementation of policies related to sustainable plastic waste management while integrating technological developments in plastic waste management strategies and investing in the infrastructure development needed for sustainable plastic waste management to achieve sustainable development.

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