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Soils in distress: The impacts and ecological risks of (micro)plastic pollution in the terrestrial environment

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ABSTRACT

Plastics have revolutionised human industries, thanks to their versatility and durability. However, their extensive use, coupled with inadequate waste disposal, has resulted in plastic becoming ubiquitous in every environmental compartment, posing potential risks to the economy, human health and the environment. Additionally, under natural conditions, plastic waste breaks down into microplastics (MPs<5 mm). The increasing quantity of MPs exerts a significant burden on the soil environment, particularly in agroecosystems, presenting a new stressor for soil-dwelling organisms. In this review, we delve into the effects of MP pollution on soil ecosystems, with a specific attention to (a) MP transport to soils, (b) potential changes of MPs under environmental conditions, (c) and their interaction with the physical, chemical and biological components of the soil. We aim to shed light on the alterations in the distribution, activity, physiology and growth of soil flora, fauna and microorganisms in response to MPs, offering an ecotoxicological perspective for environmental risk assessment of plastics. The effects of MPs are strongly influenced by their intrinsic traits, including polymer type, shape, size and abundance. By exploring the multifaceted interactions between MPs and the soil environment, we provide critical insights into the consequences of plastic contamination. Despite the growing body of research, there remain substantial knowledge gaps regarding the long-term impact of MPs on the soil. Our work underscores the importance of continued research efforts and the adoption of standardised approaches to address plastic pollution and ensure a sustainable future for our planet.

1. Introduction

The use of synthetic polymers has now become commonplace in nearly every aspect of our lives. The popularity of plastics can be attributed to a number of favourable properties. For example, they are lightweight, flexible, highly durable, inexpensive to produce and cost-effective to use. Due to these characteristics, plastics are widely employed in various sectors, including healthcare, pharmaceuticals, industries such as electronics, construction, automotive, clothing, agriculture, as well as in the manufacturing, packaging and transportation of diverse products (Surendran et al., 2023; Bouaicha et al., 2022; Joos and De Tender, 2022; Yadav et al., 2022).

Abbreviations: DOC, dissolved organic carbon; DOM, dissolved organic matter; EFSA, European Food Safety Authority; EU, European Union; FDA, fluorescein diacetate; FDAse, fluorescein diacetate hydrolase; GHG, greenhouse gas; HDPE, high density polyethylene; LDPE, low-density polyethylene; LLDPE, linear low-density polyethylene; MP, microplastic; NGS, next-generation sequencing; NOAA, National Oceanic and Atmospheric Administration; NP, nanoplastic; OTU, operational taxonomic unit; PA, polyamide; PBAT, poly(butylene adipate terephthalate); PBS, poly(butylene succinate); PC, polycarbonate; PCL, polycaprolactone; PE, polyethylene; PES, polyester; PET, poly(ethylene terephthalate); PHAs, polyhydroxyalkanoates; PHB, poly(3-hydroxybutyrate); PHBV, poly(3-hydroxybutyrate-co-3-hydroxyvalerate); PLA, poly(lactic acid); PP, polypropylene; PS, polystyrene; PTFE, poly(tetrafluoroethylene); PUR, polyurethane; PVC, polyvinyl chloride; ROS, reactive oxygen species; SBR, styrene-butadiene rubber; SI, International System of Units; SOC, soil organic carbon; SOM, soil organic matter; TPS, thermoplastic starch.

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The mass production of plastics dates back to the 1950 s. Since then, the amount of plastics produced worldwide has surged from 1.5 million tonnes per year to 374.8 million tonnes in 2019. Following a brief stagnation in 2020 (375.5 million tonnes) due to the Covid-19 pandemic, it rebounded in 2022 (390.7 million tonnes) (Bouaicha et al., 2022; Plastics Europe, 2022). This upward trend is expected to continue, with a global production projected to reach 940 million tonnes by 2040 (Surendran et al., 2023).

A significant proportion of plastic products, such as single-use plastics and packaging materials, become waste immediately after use, necessitating proper treatment (Liwarska-Bizukojc, 2021; Millican and Agarwal, 2021). In the European Union (EU) countries, the demand for plastics was nearly 50 million tonnes in 2014, of which only 54% was appropriately treated and 16% was recycled (Kumar et al., 2020). The continuous production and consumption of plastics, coupled with inadequate waste management, result in the accumulation of plastics from various sources in the environment. Approximately 79% of the 6300 million tonnes of plastic waste cumulatively generated worldwide between 1950 and 2015 end up in landfills or other environmental compartments (Surendran et al., 2023; Bouaicha et al., 2022; Geyer et al., 2017). Therefore, it is not surprising that plastics are now ubiquitous in the Earth's ecosystems as a consequence of increased human activity (Bouaicha et al., 2022; Yadav et al., 2022).

Although the problem of plastic pollution was initially described in relation to marine ecosystems (plastic waste was first found in the environment in the 1960 s, and microplastics were subsequently detected in the Sargasso Sea in 1972 and off the coast of New Zealand in 1978), terrestrial areas are estimated to experience 4–23 times more plastic pollution annually than marine waters (Zhu et al., 2023). As a result, the soil environment serves as a significant sink for (micro) plastics. However, the majority of studies conducted in recent decades have primarily focused on plastic pollution in marine environments. Consequently, our understanding of the effects of (micro)plastics on soils and terrestrial ecosystems remains extremely limited (Shafea et al., 2023; Joos and De Tender, 2022; Baho et al., 2021; Hou et al., 2021; Xu et al., 2020), and it has only recently been emphasised (Rillig, 2012).

An occurrence in which ecosystem behaviour deviates from the expected predictions is referred to as an ecological surprise (Filbee-Dexter et al., 2017). According to Doak et al. (2008), such a situation can arise, among other things, when a changing abiotic condition has a cascading effect on the distribution of certain species and their interactions. To prevent ecological surprises associated with plastic pollution of terrestrial habitats, relying solely on short-term studies of a single model organism under controlled conditions may not be sufficient. Studies that assess the consequences of soil plastic contamination over longer time scales, under realistic environmental conditions and in multi-species communities are better suited to unveil ecologically significant effects. These studies provide a more comprehensive understanding of the intricate relationships and interactions between abiotic and biotic systems. Additionally, they enable the assessment of individual species and natural communities, as well as ecosystem processes and functions, within an increasingly complex system. If these systems are compromised by environmental changes associated with plastic pollution, not only can the integrity of these complex systems be damaged, but human well-being may also be jeopardised (Baho et al., 2021).

Despite the growing number of excellent review papers on the effects of plastic pollution in terrestrial ecosystems, our understanding remains limited. Therefore, this topic continues to merit special attention. As a priority in environmental research, it is crucial to periodically consolidate the existing knowledge, incorporating the latest research findings. This review uniquely combines well-documented fundamental processes of soil science with the continually evolving research on the effects of (micro)plastics. Our main objective is to describe the general features and environmental functions of soils and thus to elucidate the changes that plastics, particularly microplastics, induce in the soil in a comprehensive and systematic approach. Firstly, we focus on the origin, occurrence, migration and environmental fate of plastics. Secondly, we aim to identify the main effects of plastics on soil parameters, with a special attention to microbial communities, enzyme activity, nutrient cycling and greenhouse gas emissions by analysing trends in (micro) plastic-induced changes in soil parameters. Additionally, our review highlights the need for further research in specific areas to acquire a comprehensive understanding of the long-term effects and potential risks of (micro)plastics in terrestrial ecosystems.

2. Plastics production and polymer types

Humans have been utilising crafted materials for thousands of years. Historical evidence indicates that birch tar was employed as an adhesive for spearheads as early as 50,000 years ago. The first industrially produced plastics emerged in the 19th century and included rubber, celluloid and viscose. Initially, these plastics were derived from bio-based sources. However, with the advent of large-scale extraction and refining of petroleum, fossil-based plastics became more prevalent (Wolter et al., 2022). The mass production of plastics further accelerated around the 1950 s (Bouaicha et al., 2022; Pérez-Reverón et al., 2022). Due to their favourable properties, the use of plastics has sparked a revolution in numerous industrial sectors, making them the most widely used synthetic materials worldwide (Wolter et al., 2022).

From a chemical point of view, plastics are relatively high-weight macromolecules formed through polymerisation, which involves the repetitive combination of smaller subunits known as monomers (Kumar et al., 2020; Hartmann et al., 2019). By modifying the monomers and the polymerisation process, it is possible to induce microstructural changes in the chemical structure of the polymer that fundamentally alter the morphology and functional properties of the plastic (*e.g.* crystallinity, gas permeability, thermal stability, polarity, mechanical properties, *etc.*), thus opening up almost limitless applications (Millican and Agarwal, 2021). The physicochemical characteristics of pure polymers can be further enhanced by incorporating various low molecular weight additives such as plasticisers, flow modifiers, stabilisers, flame retardants, antioxidants, dyes and fragrances (Zeb et al., 2023; Kumar et al., 2020; Hartmann et al., 2019). These additives can bestow valuable characteristics upon to the final product (*e.g.* increased plastic flexibility, enhanced UV resistance, reduced flammability, *etc.*), however, they can also affect the biological properties of plastics, including biodegradability and (if they are leached from the polymer matrix) toxicity (Liwarska-Bizukojc, 2021). In addition to the above, plastics can also encompass copolymers composed of multiple monomer species (*e.g.* styrene-butadiene rubber, ethylene-vinyl acetate, acrylonitrile-butadiene-styrene, *etc.*), as well as composites consisting of a polymer matrix and a (non)polymeric reinforcement (*e.g.* polyester-cotton blend textiles) (Hartmann et al., 2019).

Conventional plastics are typically petroleum-based (Table 1). The most commonly used types are polyethylene (PE) [low-density polyethylene (LDPE): homopolymer of ethylene with a densely branched structure; linear low-density polyethylene (LLDPE): copolymer of ethylene and longer-chain olefins, linear structure with short branches; high-density polyethylene (HDPE): homopolymer of ethylene with linear structure], polypropylene (PP), polystyrene (PS), polyvinyl chloride (PVC), poly(ethylene terephthalate) (PET) and polyurethane (PUR) (Hartmann et al., 2019; He et al., 2018). As an alternative to conventional plastics, there are bio-based plastics (or bioplastics), which are produced from non-fossil feedstocks. Bio-based monomers can be utilised to create both conventional (*e.g.* bio-PET and bio-PE) and biodegradable polymers, such as poly(lactic acid) (PLA), polyhydroxyalkanoates (PHAs), poly (butylene succinate) (PBS) and thermoplastic starch (TPS) (Wolter et al., 2022; Fredi and Dorigato, 2021; Hartmann et al., 2019). Nevertheless, it is important to note that not all bioplastics are biodegradable (Fredi and Dorigato, 2021; Millican and Agarwal, 2021).

Following their use, a significant proportion of plastics are incinerated, recycled or reused. However, approximately one-third of the generated waste still finds its way into the natural environment (Zhou

Table 1
Several examples of conventional and bioplastics (adapted from Wolter et al., 2022). Both bio-based and biodegradable plastics are referred to as bioplastics. However, bio-based plastics are not necessarily biodegradable.

Polimer	Plastic type	Abbreviation	Chemical formula	Biodegradability	Application	Reference
Petroleum-based	Polyethylene	PE	$[C_2H_4]_n$	No	Packaging (wrapping films, bags, bottles, sic-pack rings); straws; nettings; jugs; wire cables; toys; crates; mulch films for agriculture	Wolter et al. 2022, Wang et al. (2021a)
Petroleum-based	Polypropylene	PP	$[C_3H_6]_n$	No	Disposable surgical masks; bottle caps; ropes; nettings; plant pots; car bumpers; office supplies (folders)	Mészáros et al. 2022,Wang et al. (2021a)
Petroleum-based	Polystyrene	PS	$[C_8H_8]_n$	No	Protective packaging; food packaging (meat trays, egg holders); disposable cups and plates; carry-out containers; laboratory ware; toys	Wolter et al. 2022, Wang et al. (2021a)
Petroleum-based	Polyvinyl chloride	PVC	$[C_2H_3Cl]_n$	No	Packaging (bottles); food packaging; boots; clothing; eqipment for biomedical application; pipes	Wang et al. (2021a)
Petroleum-based	Poly(ethylene terephthalate)	PET	$[C_{10}H_8O_4]_n$	No	Packaging (bags, wrapping films, bottles, jars, tubes, blisters); food packaging; trays, textiles for clothing	Wolter et al. 2022, Wang et al. (2021a)
Petroleum-based	Polyurethane	PUR	$[C_3H_8N_2O]_n$	No	Furniture; insulation	Wolter et al. 2022
Petroleum-based	Poly(butylene adipate terephthalate)	PBAT	$[C_{12}H_{12}O_4]_m[C_{10}H_{12}O4]_n$	Yes	Packaging (wrapping films); compostable organic waste bags; disposable tableware; mulch films for agriculture	Fredi and Dorigato (2021)
Petroleum-based	Polycaprolactone	PCL	$[C_6H_{10}O_2]_n$	Yes	Sutures, drug delivery systems and tissue engineering scaffolds for biomedical application	Fredi and Dorigato (2021)
Bio-based	Bio-polyethylene	bio-PE	$[C_2H_4]_n$	No	Same applications as for PE	Wolter et al. 2022, Fredi and Dorigato (2021),Wang et al. (2021a)
Bio-based	Bio-poly(ethylene terephthalate)	bio-PET	$[C_{10}H_8O_4]_n$	No	Same applications as for PET	Wolter et al. 2022, Wang et al. (2021a)
Bio-based	Polyhydroxyalkanoates	PHAs	$[C_{10}H_8O_4R]_n$ *	Yes	Single use items in packaging; bioresorbable surgical sutures, wound dressings, bone fracture fixation plates, tissue scaffolds and porous sheets for biomedical application	Fredi and Dorigato (2021)
Bio-based	Poly(lactic acid)	PLA	$[C_3H_4O_2]_n$	Yes	Single use items in packaging; disposable cutlery, bowls, cups, bottles, jars and films; mulch films for agriculture; textiles for clothing and furnitures	Fredi and Dorigato (2021)
Bio-based	Poly(butylene succinate)	PBS	$[C_8H_{12}O_4]_n$	Yes	Shopping bags; packaking; plant pots; mulch films for agriculture	Fredi and Dorigato (2021)

*R represents CH_3 and C_2H_5 for poly(3-hydroxybutyrate) (PHB) and poly(3-hydroxybutyrate-co-3-hydroxyvalerate) (PHBV), respectively.

et al., 2021a; Kumar et al., 2020). Interestingly, during the early years of plastic production, the emphasis was on enhancing the stability and durability of conventional polymers (Millican and Agarwal, 2021). Paradoxically, these very properties are now accountable for the emergence of plastic pollution as a pressing environmental crisis.

3. Plastic pollution in the soil

3.1. Sources

Anthropogenic activities can be directly or indirectly responsible for the accumulation of plastics in terrestrial ecosystems (Yadav et al., 2022). Empirical calculations suggest that around 32% of all produced plastics potentially remain in continental environments (de de de Souza Machado et al., 2018). The substantial amount of environmentally available plastic arises from both unintentional and intentional pollution sources of pollution. Unintentional soil contamination occurs through various means, such as the application of soil amendments like compost or sewage sludge, as well as irrigation with water from plastic-contaminated water bodies. Intentional sources of plastics in the soil environment include practices such as littering, plastic mulching, irrigation with untreated wastewater and the use of plastic-containing fertilisers (Zhu et al., 2023; Joos and De Tender, 2022; Xu et al., 2020; He et al., 2018). Moreover, soil contamination with plastics can be also derived from flooding, atmospheric deposition, tyre wear or disposal of industrial and consumer waste (*e.g.* landfills) (Joos and De Tender, 2022; Baho et al., 2021).

As illustrated in Fig. 1, plastics can enter the soil through various pathways. Some of these pathways are associated with several agricultural activities. Although the use of plastics in agriculture (*e.g.* mulch film, plastic greenhouse sheeting, seedling trays, planting pots, sprinkler pipes, fertiliser packaging, *etc.*) accounts for a relatively small percentage (about 3–4%) of total human plastic consumption (Pérez-Reverón et al., 2022; You et al., 2022; Millican and Agarwal, 2021), agroplastics remain a significant source of plastic pollution in terrestrial environments (Shafea et al., 2023; Zhou et al., 2021a). Consequently, agro-ecosystems are increasingly exposed to the impacts of plastic pollution. This becomes particularly crucial in situations where sludge (also known as biosolids) from wastewater treatment plants is utilised as a soil amendment in agricultural areas. Wastewater treatment processes often leave substantial amounts of plastics in the sludge, leading to the application of 63,000–430,000 tonnes of (micro)plastic into agricultural soils annually in Europe, and a similar amount in North America (approximately 44,000–300,000 tonnes per year) (Baho et al., 2021). In these regions, sewage sludge is commonly used for soil replenishment. It is estimated that the application of biosolids can result in a 2–3 times

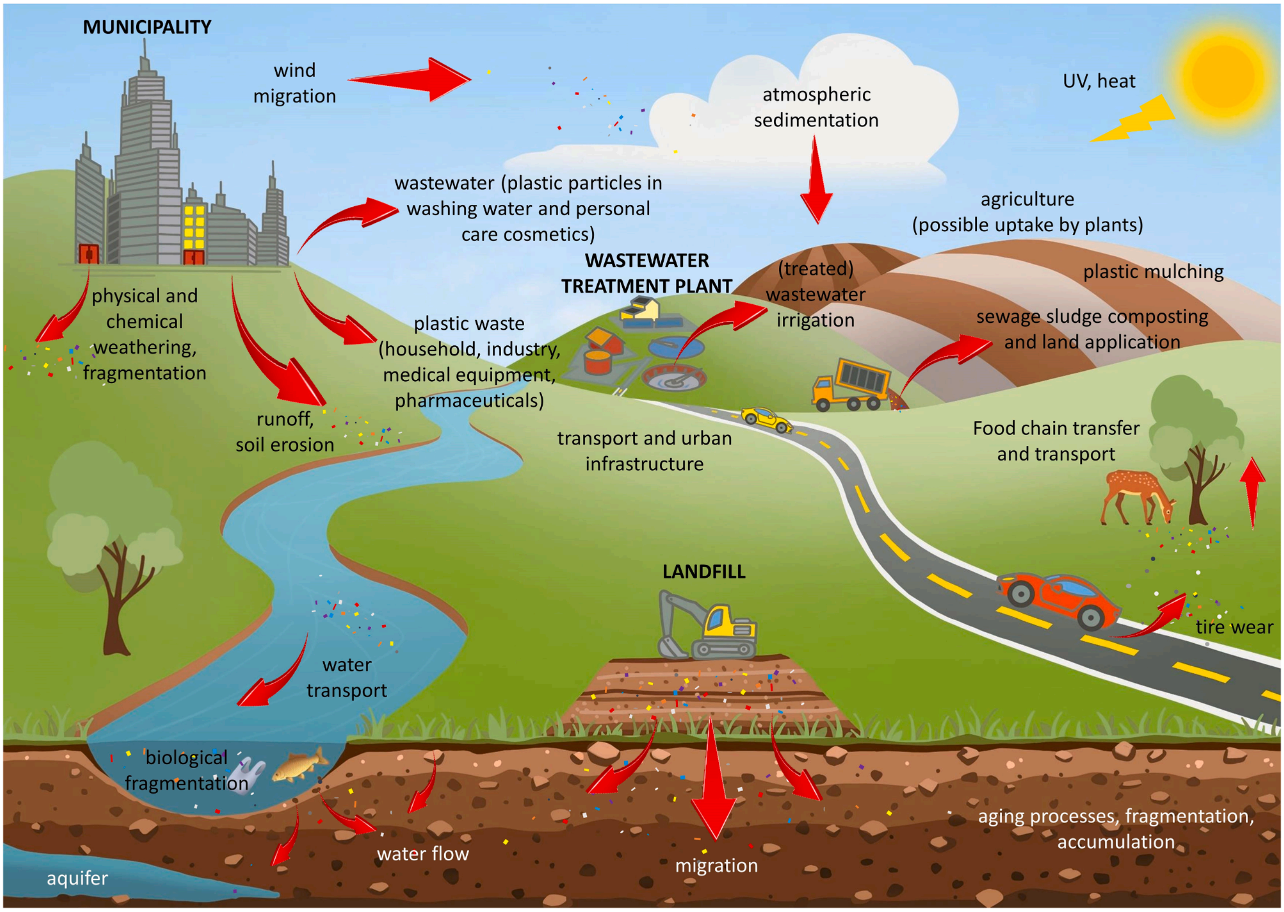

Fig. 1. Potential sources of (micro)plastics and their migration in the continental region (adapted and modified from Wang et al., 2022b, Kumar et al., 2020).

increase in the concentration of (micro)plastic in soils each year (Joos and De Tender, 2022).

Plastic mulching is a widely adopted agricultural practice, primarily utilised to enhance crop quality and yield (He et al., 2018). Three Asian countries (*i.e.* China, Japan, and South Korea) account for nearly 80% of global plastic mulch film consumption. In China, specifically, the use of mulch film has quadrupled from 0.64 million tonnes to 2.60 million tonnes per year between 1991 and 2015 (Hou et al., 2021; Xu et al., 2020). In Europe, approximately 4270 km^2 of agricultural land was covered with plastic mulch in 2010, and the practice continues to intensify with a yearly increase of 5–10% in plastic mulch usage worldwide (He et al., 2018). However, inadequate agricultural practices can leave a significant amount of mulching residue (*i.e.* plastic mulch film or its fragments) in the soil. These plastic residues are subjected to photodegradation, thermo-oxidative degradation and mechanical abrasion at the soil surface or in the soil, resulting in the continuous degradation and fragmentation of plastic debris into smaller particles (Xu et al., 2020). The majority of plastic fragments found in soil are PP (50.51%) or PE (43.43%), indicating that (micro)plastic pollution in agricultural soils is primarily associated with the use of mulch film (He et al., 2018). Land cultivation practices such as ploughing or tilling can further break down these plastic fragments, facilitating their movement into deeper soil layers. Along the larger cracks formed during tillage or through turbation by soil biota, plastic particles can reach the deeper topsoil or even the plough layer, and they can be subsequently transported by (ground)water movement (Bouaicha et al., 2022; Ren et al., 2022; He et al., 2018).

3.2. Vertical and horizontal transport

Soil-polluting (micro)plastics can move either vertically or horizontally (Fig. 1) *via* abiotic (*e.g.* wind or water) and biotic (*e.g.* human activities, plants or soil organisms) factors (You et al., 2022). The transport processes depend on the characteristics of the plastics (*e.g.* shape, size, type, density, *etc.*), but are also strongly influenced by soil properties (*e.g.* pore size, bulk density, *etc.*). For instance, spherical plastic particles can pass more easily through soil pores, while plastic fibres are more likely to be retained by soil particles. Generally, polymers with higher densities tend to migrate to deeper soil layers, while those with lower densities are transported horizontally near the soil surface by wind or water (Zhang et al., 2023a; Leed and Smithson, 2019). Aeolian transport of (micro)plastics can be significant, particularly in arid and semi-arid regions. Particles with diameters of less than 100 μm can be carried in the air and subsequently deposited onto soils (through either dry or wet deposition) (Pérez-Reverón et al., 2022; Rezaei et al., 2022). Infiltrating water can result in the denser and smaller plastic particles to leach deeper into the soil profile (Zhang et al., 2023a; O'Connor et al., 2019). During surface runoff, plastics can be trapped by plant stalks, shoots and leaves, thereby reducing their mobility (Han et al., 2022). At the root level, vegetation also plays a crucial role in influencing the vertical distribution of plastics in the soil. Primary and secondary roots increase soil porosity, allowing less dense plastic particles to float to the surface when pore spaces are filled with water. Plants with tertiary roots tend to retain the plastics in the soil layer (Li et al., 2021a). Soil animals (*e.g.* springtails, earthworms, digging mammals) also contribute to the transfer of (micro)plastics within soils (Dissanayake et al., 2022a; Bouaicha et al., 2022; Lwanga et al., 2017, 2016). Transport processes associated with water and wind enable plastics to travel longer distances, eventually reaching larger freshwater bodies and oceans. On the other hand, common farming techniques (*e.g.* tillage, digging, raking, ploughing and irrigation) and bioturbation by soil-dwelling organisms primarily contribute to the short-distance movement of plastics (Pérez-Reverón et al., 2022; Zhu et al., 2019). Given that (micro)plastics can adsorb various organic/inorganic compounds (*e.g.* heavy metals, pesticides, hydrocarbon derivatives), the combined transport of plastic particles and secondary pollutants can have significant environmental consequences in both aquatic and terrestrial ecosystems, emphasising the need for careful attention (Sahai et al., 2023; Zhu et al., 2019).

3.3. Deterioration and ageing

Topsoil provides a potentially degradative environment for (micro) plastics due to direct UV exposure, increased oxygen availability and relatively high temperature (Zhu et al., 2023). However, (micro)plastic degradation (especially in deeper soil layers) either does not occur or happens extremely slowly under natural conditions (Xu et al., 2020; He et al., 2018). Studies has shown that for PE and PP, only 0.1–0.4% and 0.4% weight loss were measured after 800 days and 1 year of incubation in soil, respectively, while no degradation was observed for PVC after 35 years (Zhu et al., 2019; He et al., 2018). This also indicates that the pedosphere (particularly agricultural soils) acts as a final sink for (micro)plastics (Hou et al., 2021), with their amount in the soil not decreasing (or decreasing only minimally) and even continuously increasing due to ongoing plastics inputs. Despite the limited degradation of (micro)plastics in soil, their physical and chemical properties can undergo several changes (Table 2) when exposed to abiotic and biotic factors (*e.g.* sunlight, water corrosion, sand friction, ingestion by soil fauna, *etc.*), a process known as plastic ageing (Bouaicha et al., 2022; Miranda et al., 2021). Under natural conditions, UV radiation, thermal oxidation, abrasion and biodegradation can alter the chemical structure of the polymer by promoting the cleavage of intramolecular bonds, the formation of cross-links or an increase in the presence of oxygen-containing functional groups (Zeb et al., 2023; Tian et al., 2022; Mao et al., 2020; Zhu et al., 2019). These changes can lead to the release of additives and adsorbed secondary contaminants, as well as a decrease in the hydrophobicity, tensile strength and molecular weight of the plastic polymer. Additionally, surface roughness, microcracks, polarity, carbonyl index and sorption capacity may increase. Ultimately, these processes can result in a reduction in particle size (Bouaicha et al., 2022; Miranda et al., 2021; Ren et al., 2021; Mao et al., 2020). Furthermore, the fragmentation taking place during ageing processes can promote the microbial colonisation and biofilm formation on the surface of plastic debris (Rillig et al., 2023; Bouaicha et al., 2022), creating a new ecological niche known as the plastisphere, a human-made microbial ecosystem (Wang et al., 2022a; Amaral-Zettler et al., 2020).

Table 2
The physicochemical changes of microplastics (MPs), which alter their characteristics, can vary with increasing ageing degree ('+' and '-' indicate an increasing and decreasing effect, respectively).

Physicochemical processes and characteristics	Trend of change during ageing
Cleavage of intramolecular bonds	+
Cross-linking	+
Functional groups	+
Oxygen-containing functional groups	+
Carbonyl index	+
Adsorbed secondary contaminants	+
Released additives	+
Hydrophobicity	-
Polarity	+
Contact angle	-
Tensile strength	-
Molecular weight	-
Surface roughness	+
Microcracks	+
Porosity	+
Specific surface area	+
Sorption capacity	+

3.4. Macro- and mesoplastics

As a result of continuous fragmentation, plastic particles of various sizes occur simultaneously in the polluted environments (Zeb et al., 2023; Xu et al., 2020). These fragments can be classified based on different factors (*e.g.* chemical composition and structure, shape, colour, solubility, origin, *etc.*), but the most common categorisation is based on the size range they fall into (Jain et al., 2023; Bermúdez and Swarzenski, 2021).

According to the most widely used nomenclature, plastic fragments larger than 25 mm are referred to as macroplastics. In China, macroplastic contamination is estimated to be around 10–100 kg per hectare (up to 10 million items/ha), but decades of plastic mulching may lead to much higher contamination (Xu et al., 2020).

Smaller fragments in the size range of 5–25 mm are known as mesoplastics. Compared to even smaller plastic particles, mesoplastics can be present in the environment in lower numbers but at significantly higher mass concentrations. For instance, in one kilogram of floodplain soil, the detected range for mesoplastics was 0–89 pieces, while the corresponding mass ranged from 0 to 295 mg. In comparison, the detected range for smaller plastic particles was 0–593 pieces, with a range of approximately 0–56 mg in mass (Xu et al., 2020; Scheurer and Bigalke, 2018).

3.5. Microplastics

With the further fragmentation of macro- and mesoplastics, even smaller particles are constantly formed. Thus, soils heavily contaminated with macro- and mesoplastics are almost certain to contain so-called microplastics (MPs). MPs are defined as plastic particles with a diameter of less than 5 mm (Xu et al., 2020). MPs can be further categorised based on their size into three groups: large MPs (3–5 mm), medium MPs (1–3 mm) and small MPs (<1 mm). However, the European Food Safety Authority (EFSA) provides a slightly more precise definition, considering MPs to have a size range between 0.1 μm and 5000 μm. Additionally, plastic particles smaller than 0.1 μm in diameter are referred to as nanoplastics (NPs) according to EFSA's classification (Bouaicha et al., 2022).

The term "microlitter" was first used in 2003 to describe marine plastic litter within the size range of 63–500 μm. Subsequently, the term "microplastic" gained popularity in 2004, but at that time it was used to refer to plastics that were truly microscopic, measuring less than 20 μm. The current definition (MPs<5 mm) was only widely adopted following a meeting of the National Oceanic and Atmospheric Administration (NOAA) in 2008. However, it should be noted that this definition is ambiguous and subject to controversy in the scientific community due to the dimensions involved (Hartmann et al., 2019). There is now a growing demand for a more consistent and comparable nomenclature in the field. Researchers such as Bermúdez and Swarzenski (2021) advocate for clarifying and expanding the four basic categories of plastics (*i.e.* macro, meso, micro and nano) based on the International System of Units (SI). They also emphasise the need for a new terminology (*e.g.* micro-sized plastic instead of microplastic) to avoid misunderstandings in future research.

MPs are not solely generated through the fragmentation of larger plastics. Based on their origin, they can be classified into two categories: primary and secondary MPs (Shafea et al., 2023; Dissanayake et al., 2022a). Primary MPs are intentionally produced for commercial purposes or as industrial precursors for other products. Examples of primary MPs include plastic pellets, microbeads (commonly found in cosmetics and skin scrubs), industrial abrasives and other consumer products. On the other hand, secondary MPs are released into the environment as a result of the fragmentation and ageing of larger plastics (or even primary

MPs). The structural integrity of plastic particles deteriorates due to physical, chemical, biological or photodegradation processes, leading to their reduction in size, sometimes becoming undetectable to the naked eye (Prabhu et al., 2022; Yadav et al., 2022). Typical examples for secondary MPs include fibres released during the washing of synthetic clothing or particles from the abrasion of plastic coatings and vehicle tyres (Ullah et al., 2021).

MPs can also be classified based on their polymer composition. The polymers detected in environmental samples correspond to the most commonly used plastic types such as PE, PP, PVC, PS, PET, PUR, polyamide (PA), styrene-butadiene rubber (SBR) (Wang et al., 2022b; Yang et al., 2021a). The prevalence of specific polymers varies across different regions. For instance, in India, Germany and China, MP pollution is primarily associated with PE and PP. PE dominates the majority of MPs found in India, while PP is more abundant in Spain (You et al., 2022). PVC was detected in the highest proportion (80%) in soil samples from a waste facility in Sydney, Australia (Fuller and Gautam, 2016). Although PE and PP are the most prevalent polymer types in arable soils, PET and PVC can also originate from agricultural sources (You et al., 2022; Yang et al., 2021a). In China, where plastic mulch film consumption reaches 2.6 million tonnes per year (approx. 30% of global consumption), plastic mulch residues contribute significantly to MP pollution. In contrast, MP pollution in Europe and North America is primarily attributed to agricultural sewage sludge disposal, which can load soils with 63,000–430,000 tonnes and 44,000–300,000 tonnes of MPs, respectively (You et al., 2022). It is important to note that the increased use of disposable surgical masks and protective equipment made of PP during the Covid-19 pandemic has further exacerbated environmental burdens and serves as a source of secondary MPs (Cabrejos-Cardeña et al., 2023; Mészáros et al., 2022; Prabhu et al., 2022).

MPs exhibit a wide range of shapes, reflecting their diverse origins and sources of contamination. They can appear as beads (with spherical shapes), films (forming membranes), fragments, fibres, irregular ovoid particles, foam or pellets (resembling short rods) (Wang et al., 2022b; Baho et al., 2021; Yang et al., 2021a). MP fibres, for example, are commonly found in soils treated with sewage sludge and subjected to irrigation with wastewater or contaminated running water, while MPs that has been exposed to environmental conditions over extended periods are more likely fragments (Pérez-Reverón et al., 2022.Wang et al., 2022b).

The shape, size and chemical composition of MPs are not only important factors in their classification, but play crucial roles in determining their effects on soil properties and the associated environmental risks. According to Rillig et al. (2019), MPs with shapes differing significantly from soil particles can exert stronger effects on soil. Thus, MP fibres and films are suggested to have a greater impact on soil properties compared to spherical MPs like beads and particles, which closely resemble soil particles. This theory has been supported by subsequent studies (Lehmann et al., 2021; Lozano et al., 2021a). The chemical composition of MPs, including the high-molecular-weight polymer and the presence of additives and adjuvants, determines their density, degradability and toxicity, thereby influencing soil bulk density, as well as soil chemical and biological properties. Smaller MPs (particularly NPs) have a tendency to clog soil micropores, but they also pose a greater risk of entering biological systems, presenting a significant environmental challenge. Additionally, due to their small size, larger specific surface area and their generally hydrophobic surface, smaller MPs can act as vectors for other pollutants (*e.g.* heavy metals, persistent organic pollutants, pesticides, pharmaceutical toxic substances, flame retardants and plasticisers). They can adsorb and transport these pollutants, thereby facilitating their dispersion in the environment (Sahai et al., 2023; Wang et al., 2022b). As MPs exhibit exceptional persistence in the environment, they can be transferred through the food chain, accumulating in plants and animals and eventually reaching humans as the final consumers (Othman et al., 2021).

The presence of MPs has been confirmed in all tested soil types, whether agricultural, industrial, urban or uncultivated, indicating that soil is a major reservoir for MPs. However, there are notable variations in their distribution, composition, size and shape. These differences can be attributed to various potential sources of MPs (*e.g.* plastic mulching in agricultural soils, tyre dust and road paints in traffic soils), research objectives, and the limitations of current testing and quantification methods (Wang et al., 2022b). In general, agricultural soils tend to be more contaminated with MPs through sewage sludge and compost-based fertilisation, plastic mulching or wastewater irrigation. Forest soils and uncultivated urban or industrial soils, on the other hand, receive MPs mainly through atmospheric deposition (Fig. 1) (Zhu et al., 2019).

It is evident that MPs are highly heterogeneous pollutants, consisting various polymers, occurring in diverse sizes and shapes, and potentially containing a wide range of additives and adsorbed contaminants. Unlike other chemical pollutants, MPs possess several unique characteristics: (a) they are persistent and can persist in the environment for centuries, (b) they exhibit complex interactions with the abiotic environment, (c) they have direct and indirect effects on terrestrial organisms, (d) they facilitate the movement of other contaminants by interacting with them. Considering these factors, it can be concluded that compared to other pollutants, MPs are more likely to have significant effects on polluted media and ecosystems, thus, their presence in the environment increases the likelihood of ecological surprises (Baho et al., 2021).

4. Soil functions and the potential risks of soil microplastic pollution

Soil is part of the natural environment and performs pivotal ecosystem functions. These functions include (a) decomposing and transforming organic matter and toxic compounds, (b) providing nutrients and water for plants, (c) purifying water, (d) controlling pests and diseases, or (e) mitigating greenhouse gases (Hartmann and Six, 2023). Soil, also known as the pedosphere, is the outermost solid and fertile layer of the Earth's crust. It receives, partially stores and transforms material and energy fluxes, ensuring the cycling of materials in biological systems (Stefanovits et al., 1999). Thus, soil enables the supply of nutrients to plants, which in turn leads to the production of primary biomass, making agroecosystems globally essential ecological and economic networks with direct impacts on both climate and human nutrition (Hartmann and Six, 2023; Foley et al., 2005). In addition, as a complex system of mineral and organic components forming a network of water- and air-filled pores, soil also provides a heterogeneous habitat for a variety of soil-dwelling organisms that drive key soil functions. Macrofauna (*e.g.* earthworms) plays a role in breaking down larger organic materials, (re)distributing smaller pieces and increasing their availability. This process enhances microbial colonisation due to the increased surface area (Joos and De Tender, 2022). Microorganisms are responsible for degrading organic matter and converting it into nutrients available for plants *via* oxidation, reduction, chelation and solubilisation (or storing it in the dead biomass, also known as the necromass) (Hartmann and Six, 2023). Soil microbiota (*i.e.* bacteria, fungi, archaea, viruses and protista) mediate 80–90% of the soil processes. Microbial diversity is an essential driving force for these processes, as microorganisms directly contribute to carbon sequestration in the soil, nutrient cycling and soil structure formation (*i.e.* by incorporating organic matter from the microbial decomposition processes) (Joos and De Tender, 2022; Kibblewhite et al., 2008; Nannipieri et al., 2003). Soil physical, chemical and biological properties are therefore not independent from each other: although the composition and the functions of the soil microbiota can influence soil physicochemical parameters, reversely, these parameters can also determine microbial activity and survival. Changes in the soil structure have a significant impact on soil-dwelling microorganisms, which, in turn, affect soil organic matter (SOM), nutrient cycling and microbe-plant interactions (Hartmann and Six, 2023; Six et al., 2006).

Soil structure refers to the three-dimensional, porous arrangement created through the integration of minerals and organic matter into aggregates of varying sizes (Hartmann and Six, 2023). The pore space is filled with air and/or water, carrying dissolved nutrients, thus providing a habitat for microorganisms (Joos and De Tender, 2022). Within the soil environment, the majority of microbial assemblages are associated with soil particles, particularly microaggregates, rather than macroaggregates. Consequently, the porous soil structure fundamentally determines the resources accessible to these microorganisms (*e.g. via* water flow, oxygen diffusion, availability and accessibility of nutrients or organic matter). The structure, size and stability of soil aggregates are therefore key factors influencing the abundance, diversity and activity of soil-dwelling microorganisms, thereby impacting plant development, soil erosion and the turnover of organic matter (Hartmann and Six, 2023).

The soil system relies on a diverse array of microorganisms to carry out essential processes, making it resilient to the loss of a few species. This functional redundancy is what contributes to the robustness of terrestrial ecosystems (Joos and De Tender, 2022; Jurburg and Salles, 2015). However, it is important to avoid pushing the limits of soil buffering capacity too far, as excessive exploitation can lead to soil degradation (Kopittke et al., 2019). Anthropogenic impacts stemming from intensive agriculture, pollution or global warming induce changes in soil structure, as well as in soil physical and chemical properties. As a result, these alterations can have profound effects on the diversity and functions of soil microbiota, ultimately disrupting the dynamics of the soil food web (Joos and De Tender, 2022).

In light of the above, a thorough risk assessment of MP contamination in soils is crucial to evaluate the potential degradation of soil functions. Nonetheless, it is important to acknowledge that making accurate comparisons of the MP impacts on soil physicochemical and biological parameters is highly challenging and often subject to controversy due to significant variations in (a) the testing methods applied, (b) the characteristics of the plastic materials tested (*e.g.* size, shape, concentration *etc.*) and (c) the diverse soil conditions under study (*e.g.* soil type, moisture content, experimental scale, time interval *etc.*).

4.1. Microplastic effects on soil physical properties

MP accumulation in the soil has a significant impact on various soil physical properties, including bulk density, size and distribution of water stable aggregates, porosity and water dynamics such as infiltration, retention and evaporation. However, these alterations are not uniform and are highly dependent on factors such as soil type, climatic conditions, as well as the abundance, parental material and shape of MPs (Baho et al., 2021; Iqbal et al., 2021). For instance, the presence of films, fibres and foams/fragments can decrease soil bulk density, increase water holding capacity and decrease soil aeration/porosity, respectively (Lozano et al., 2021a).

4.1.1. Aggregation

Soil aggregates, which consist of particles of varying sizes typically ranging from 2 to 200 μm in diameter, determine soil structure, pore size and stability, thereby regulating water flow (and hence nutrient distribution), soil aeration, erosion susceptibility and soil microbial activity (Wang et al., 2022b; Yadav et al., 2022). MPs are often incorporated into soil aggregates. In fact, a study by Zhang and Liu (2018) found that 72% of plastic particles were associated with soil aggregates, while only 28% were dispersed. Specifically, microfibres have been observed to be more tightly integrated into soil clumps compared to MP fragments, and they also have the potential to bind soil particles together, forming larger aggregates (Wang et al., 2022b; Yadav et al., 2022). Nonetheless, the incorporation of micro-sized plastics, such as PA, PP, PET, PUR, PS, polycarbonate (PC) and polyester (PES) in various forms (fibres, foam, film or fragments), can increase the number of potential fracture points in soil aggregates, thus reducing their stability (Wang et al., 2022b; Iqbal et al., 2021). While negative effects on aggregate water stability at higher temperatures have been observed for PES microfibres (Liang et al., 2019), contradictory effects were found during drought conditions, where PES microfibres improved soil aggregation (Lozano et al., 2021b). The underlying reasons for these observations are not yet fully understood, but it can be assumed that MP pollution inevitably leads to changes in soil structure and aggregate formation, with MP shape being a major determinant. However, these effects on soil physical properties can be modified by soil biota and SOM (Wang et al., 2022b; Iqbal et al., 2021).

4.1.2. Porosity

Soil porosity plays a critical role in regulating the flow of water and soil aeration, which in turn influences the distribution of aerobic and anaerobic microorganisms, as well as the uptake of water and nutrients by plants. Surface water drains rapidly through larger pores (macropores$>$30 μm), while smaller pores (micropores$<$30 μm) tend to retain water within the soil (Mbachu et al., 2021). The impacts of MPs on soil porosity depends in particular on their shape. MPs in the form of fibres, foam and fragments can increase porosity by creating additional pore spaces within the soil matrix, hence improving soil aeration and facilitating root penetration. Other shapes, such as pellets, beads or particles, have the potential to easily clog the pore space, restricting water and air movement in the soil (Wang et al., 2022b).

4.1.3. Bulk density

Soil bulk density is an important parameter that affects soil quality, porosity, plant root development and ultimately soil fertility. When MPs (*e.g.* PE, PET, PP, PS, PES) accumulate in the soil, they typically have lower density than soil particles. As a result, MP contamination reduces soil bulk density while increasing porosity (Wang et al., 2022b). However, it is worth noting that different studies may yield various results. For example, Zhang et al. (2019a) observed no detectable changes in the bulk density of a clayey soil when 0.1% and 0.3% PES microfibres were present. Nevertheless, they did find an increase in the volume of macropores and the amount of water stable aggregates.

4.1.4. Soil moisture

Soil moisture content regulates the survival and reproduction of soil-dwelling organisms and plants by influencing the bioavailability of nutrients and pollutants. However, MP contamination can disrupt this system and lead to imbalances. Most MPs are hydrophobic, which directly impacts water retention and accessibility in the soil. Additionally, they indirectly modify several transport processes by causing changes in soil structure (Wang et al., 2022b). In a five-week garden experiment conducted by de Souza Machado et al. (2018), PA beads and PE fragments did not exhibit clear trends with increasing concentrations up to 2%. Nevertheless, PES fibres increased the water holding capacity of loamy sand soil possibly due to the fact that the fibre shape (unlike other MP shapes) is very different from the shape of soil particles (Wang et al., 2022b). Similar findings were reported by de Souza Machado et al. (2019) and Lozano and Rillig (2020), confirming a (concentration-dependent) positive effect of PES microfibres on soil water retention and plant growth. This effect was more pronounced than that of HDPE, PET or PS MPs. MP accumulation, such as PES fibres and PE films, can form water channels in the soil profile due to their lower permeability compared to soil particles. These channels facilitate faster water flow between pores and increase the rate of evaporation, thereby altering the water cycle and soil moisture (Bouaicha et al., 2022; Iqbal et al., 2021). Smaller-sized MPs and higher abundance can result in greater water evaporation. Larger plastic pieces (5–10 mm) can cause soil cracks with similar effects, which not only remove water from deeper soil layers but also facilitate the vertical transport of MPs and other contaminants, allowing them to reach the groundwater zone more rapidly (Bouaicha et al., 2022; Wang et al., 2022b). Soil-integrated MPs disrupt water dynamics, leading to water-limited conditions.

Furthermore, their impermeable nature creates a physical barrier that hinders plants from accessing water and nutrients (Iqbal et al., 2021).

Consequently, the presence of MPs in soil is anticipated to have a detrimental impact on soil erosion due to the inevitable alteration of its physical properties. Moreover, the modified hydrological conditions resulting from these alterations may exacerbate drought events. With the progression of climate change, droughts are expected to become more frequent, further compounding the negative effects of MPs on soil erosion (Zhou et al., 2021a).

4.2. Microplastic effects on soil chemical properties

4.2.1. Carbon content

Indeed, the polymers constituting MPs are primarily composed of carbon (C), and can thus potentially serve as a source of C for the soil ecosystem (Qiu et al., 2022). Nevertheless, most plastics are inert and exhibit limited or no decomposition (Zhang et al., 2021), resulting in the long-term storage and accumulation of this C content in the soil (Qiu et al., 2022). As a consequence, MPs generally do not have a direct impact on soil nutrient and element cycling (Zhang et al., 2021).

Plant nutrient availability, microbial activity and soil fertility are closely interconnected with the quantity of organic matter present in the soil. Among these organic matters, dissolved organic matter (DOM) plays an important role in soil biogeochemical cycles by facilitating the circulation of total SOM and the transfer of inorganic macronutrients (such as nitrogen and phosphorus) as well as contaminants (Ren et al., 2022; Wang et al., 2022b). However, the presence of MPs can disrupt these processes both in the soil and in the rhizosphere (Bouaicha et al., 2022). Certain microorganisms possess the ability to degrade plastics and utilise their monomers and degradation products as organic C sources, particularly in the case of bioplastics. This can potentially lead to an increase in the amount of dissolved organic carbon (DOC) in environments heavily contaminated with MPs (Ren et al., 2022; Wang et al., 2021b). Nevertheless, the effects of conventional plastics are often inconclusive. For example, PP-MPs were found to increase DOC only at a higher concentration (28%) (Liu et al., 2017), while PE-MPs (at 5% concentration) caused no significant changes in DOC (Ren et al., 2020) or even led to a decrease in SOM (Dong et al., 2015), similar to the effects observed with PS-MPs and poly(tetrafluoroethylene)-MPs (PTFE-MPs) (both applied at 0.25% and 0.5%) (Dong et al., 2021).

It is important to note that the rhizosphere contains a higher concentration of organic matter (*e.g.* biomass or compounds released by plant roots and microorganisms) compared to the bulk soil, and is present in smaller particles due to accelerated decomposition and ageing processes. Additionally, these particles in the rhizosphere tend to be more aggregated, thanks to the presence of microbial and plant-derived exopolysaccharides that act as binding agents (Bouaicha et al., 2022; Zhou et al., 2021a). The complex nature of this environment can complicate the interactions between soil, microbes, plants and MPs.

4.2.2. Nutrient content

Most plastics contain little or no nitrogen (N) and phosphorus (P), thus although their degradation increases the C:N ratio in the soil due to the C content released (or dissolved in leachate) (Bouaicha et al., 2022), they only affect inorganic nutrient levels through indirect mechanisms (Iqbal et al., 2021). In natural soils, the nutrient content is primarily derived from the decomposition of soil minerals and organic compounds, while agricultural soils are supplemented with fertilisers (Wang et al., 2022b). The effects of MPs on soil N and P can vary and have been a topic of debate. Some studies have reported negligible effects, with *e.g.* PE, PES and PLA (Bouaicha et al., 2022; Wang et al., 2022b). On the other hand, other types of MPs like PVC, PS, PTFE and PE have been associated with negative impacts on N and P levels (Wang et al., 2022b; Xu et al., 2020). Conversely, PP, PLA have shown positive effects (Yin et al., 2023). PLA can increase NO_3^--N concentration while decreasing NH_4^+-N content (Wang et al., 2022b). In the case of agricultural soils, Yan et al. (2021) found that the effect of PVC-MPs (at $\leq 1\%$) on available P varied depending on soil type, with an increase observed in acidic red soil and a decrease in neutral paddy soil Their findings also suggest that the mobility of inorganic P and the mineralisation of organic P can be influenced by the plasticiser (*e.g.* phthalate) present in the plastics used during plastic production.

Given the conflicting findings, it is likely that the direct biodegradation of MPs does not have a significant impact on soil nutrient cycling. Instead, it is the degradation of humus-like materials that appears to play a role in this process. However, MPs can still influence the circulation of soil nutrients through their effects on microbial activity (Zhang et al., 2021), as indicated by shifts in soil enzyme activities (*e.g.* urease, phosphatase), which regulate nutrient availability (Wang et al., 2022b). While the precise molecular mechanisms underlying the influence of MPs on nutrient biogeochemical cycling are not yet well understood, it can be concluded that MP pollution, particularly from sources like plastic film residues, diminishes soil fertility and inhibits healthy plant growth (Xu et al., 2020). Additionally, MP pollution contributes to the emission of greenhouse gases such as CO_2, CH_4 and N_2O (Zhuanxi et al., 2023; Inubushi et al., 2022; Ren et al., 2022; Wang et al., 2022b).

4.2.3. Co-occurring contaminants

The aforementioned effects can be further influenced by the ageing process of MPs, which can lead to their involvement in the adsorption, transportation and desorption of contaminants from a variety of external sources (*e.g.* herbicides, pesticides, persistent organic pollutants, polycyclic aromatic hydrocarbons and heavy metals) (Dissanayake et al., 2022a; Sajjad et al., 2022; Zhang et al., 2021). Under environmental conditions, the surface of MPs undergoes roughening, cracking and the development of functional groups, which create a multitude of active binding sites that facilitate the adsorption of organic compounds. Additionally, the increased specific surface area, porosity and polarity of aged MPs promote the binding of heavy metals (Ren et al., 2022). Furthermore, as MPs continue to age, the additives used in plastic production (*e.g.* plasticisers, flame retardants, antioxidants, pigments, *etc.*) as well as the adsorbed substances can be released from the polymer matrix, along with cleaved plastic monomers. This release of substances can have detrimental effects on soil ecosystem functions (Zhang et al., 2021). In addition to its impact on contaminant adsorption and desorption, MP ageing has the potential to alter soil pH through various mechanisms, including (a) the mobilisation of compounds, (b) changes in cation exchange capacity, (c) the release of protons, and (d) the formation of hydroxyl anions (Bouaicha et al., 2022; Wang et al., 2022b; Zhang et al., 2021).

4.2.4. Soil pH

Soil pH is a critical abiotic factor that determines, among other things, mineral binding capacity, soil organic carbon (SOC), nutrient and pollutant bioavailability, microbial community structure and activity, plant growth, and hence crop yield (Wang et al., 2022b). However, the impact of increased MPs presence on soil pH is still not fully understood based on the currently available data (Bouaicha et al., 2022). Some studies have shown that certain MPs, such as 1% and 10% of PLA and HDPE (Yang et al., 2021b), LDPE film, bioplastic mulch film (Qi et al., 2020), 0.4% PES fibre (Lozano et al., 2021b), 0.4% PS foam and 0.4% PET fragment (Zhao et al., 2021) can increase soil pH. In contrast, other MPs, such as 0.5% PS, 0.5% PTFE (Dong et al., 2021), 0.2% PE (Li et al., 2021b) and 0.1% HDPE, have shown the opposite effect. Additionally, 0.001% PES and 0.01% PLA did not have a significant impact on soil pH after a 30-day period (Boots et al., 2019). An acidic environment with low pH can accelerate the ageing of MPs, leading to increased release of secondary pollutants and organic compounds. Although this process can pose a risk to the natural environment, it may also compensate for the loss of SOC caused by MPs. The conflicting results suggest that not only the type and abundance of MPs, but also their size, shape, soil type, fertilisation history and the duration of the experiment can influence the effects of MPs on soil pH (Wang et al., 2022b).

4.3. Microplastic effects on soil biology

Soil provides an extremely complex habitat for a diverse array of organisms. MPs not only act as contaminants within this system by (a) accumulating in the food chain through uptake or ingestion, (b) adsorbing secondary contaminants, or (c) containing harmful additives, but they also have a direct impact on soil-dwelling organisms by altering their habitat (Kaur et al., 2022). The presence of MPs has been found to modify the physical and chemical properties of the soil, which potentially alter biodiversity, biological activity and plant health. These factors are important biological indicators of soil quality and health, as they are closely interconnected with one another (Mbachu et al., 2021).

Soil biological parameters serve as valuable ecotoxicological indicators, allowing for the assessment of how MP contamination affects the structure and functioning of terrestrial ecosystems and the associated risk to living organisms (Gruiz et al., 2001). The detection of specific responses, known as biomarkers, in test organisms exposed to chemicals released into the environment is an increasingly common practice in ecotoxicology. Various methods are available to determine these biomarkers, including the measurement of changes in enzyme activity, reproductive capacity or other physiological processes (*e.g.* respiration, relocation, feeding behaviour, *etc.*) (Connell et al., 2009). However, despite a growing body of research, there is still a considerable uncertainty surrounding the precise effects of MPs on soil biota, primarily due to limited knowledge about the interactions between soil organisms and MPs (Wang et al., 2019).

4.3.1. Microplastic effects on terrestrial vegetation

Healthy soils are pivotal in achieving sustainable food security, mitigating climate change and conserving biodiversity. Plants contribute to these goals by maintaining a healthy soil ecosystem. When faced with pollution, plants employ two main strategies to cope with pollutants: (a) avoiding uptake or (b) undergoing detoxifying enzymatic reactions, such as oxidation, reduction and conjugation, to neutralise their effects (Wang et al., 2019). The sensitivity of plants to plastic pollution is primarily determined by their ability to counteract both indirect (*e.g.* changes in soil physicochemical properties) and direct effects (*e.g.* bioaccumulation or acting as a physical barrier) (Zhang et al., 2022). The latter is particularly important as land plants possess a variety of structural elements and properties (*e.g.* root system, vascular tissues, vacuoles, cytoplasm, transpiration, plasma membrane potential, *etc.*) that can facilitate the uptake and storage of MPs (He et al., 2018).

The field of plants and MPs raises crucial questions, including: (a) the impact of MPs on plant development, (b) the potential for plants to accumulate MPs, (c) the mechanisms by which MPs subsequently enter the food chain (Iqbal et al., 2023; Mészáros et al., 2023; Kumar et al., 2020).

By altering certain soil physical parameters (*e.g.* increasing soil porosity and aeration) and soil enzyme activities (*e.g.* phosphatase, urease), MPs can enhance water and nutrient availability for plants, thereby promoting the growth of belowground plant biomass (Qiu et al., 2022). In contrast, MPs can also accumulate around root hairs, adhere to the root surface and block pores within the seed capsule, thus acting as a physical barrier to water and nutrient uptake and plant respiration. This can result in delayed seed germination and reduced plant growth (Zhang et al., 2022). In a study investigating the effect of micro-sized PP surgical mask fragments on plant development, Mészáros et al. (2022) also observed a concentration-dependent change in the number and density of lateral roots in 14-day-old oilseed rape (*Brassica napus* L.) grown in a rhizotron system (increased with 0.5% PP and decreased with 1% PP), indicating a stress-induced morphological response to MPs.

Due to their size, larger MPs cannot pass directly through the plant cell wall. However, it is now evident that some NPs can. In an experiment conducted by Li et al. (2020), the tested PS beads has a larger diameter (0.2 μm) compared to both the cell wall pores (3.5–5 nm) and the intercellular plasmodesmata (50–60 nm). Yet, they were able to penetrate and accumulate in the roots of wheat (*Triticum aestivum* L.) and lettuce (*Lactuca sativa* L.) by breaking through the Casparian strip at the formation sites of lateral roots. PS particles of 20 and 40 nm were also found to enter tobacco (*Nicotiana tabacum* L.) BY-2 cells (Bandmann et al., 2012). Therefore, the size of MPs (particularly NPs) is an important factor for their uptake, with the smaller the particles being more easily taken up by plants (You et al., 2022). Transpiration flow plays a key role in the uptake and translocation of plastics in plants (Azeem et al., 2021). These processes can occur through apoplastic or symplastic pathways. Apoplastic transport involves the movement of NPs through extracellular spaces and cell walls, eventually transitioning to the symplastic pathway by crossing the Casparian strip until reaching the vascular system. Symplastic entry of NPs into plants can occur through various mechanisms, including (a) endocytosis, (b) movement through a system of passages established by plasmodesmata and radicle cell membranes, (c) movement through pores created by NPs on the plasma membrane and (d) movement through membrane channel proteins (Campanale et al., 2022). Aquaporins, for example, facilitated the entry of PS-NPs into the roots of rice (*Oryza sativa* L.) (Zhou et al., 2021b). While the root system is one of the primary entry points for NPs, they can also enter plants through the stomata on the leaf surface during atmospheric deposition (Campanale et al., 2022).

The presence of NPs can then trigger various physiological and biochemical responses (*e.g.* reductions in biomass, height or leaf area, altered photosynthetic efficiency, reduced pigment content, *etc.*) (Campanale et al., 2022). Foliar exposure of lettuce (*L. sativa*) to PS-NPs increased the amount of reactive oxygen species (ROS), leading to significant stress responses and a decrease in total antioxidant capacity (Lian et al., 2021). In rice (*O. sativa*), PS-MPs (<50 μm) affected energy metabolism and the rate of anabolism, compromising plant quality and nutritive value (Wu et al., 2020). These effects can inhibit seed germination [PS: garden cress (*Lepidium sativum* L.), PLA: perennial ryegrass (*Lolium perenne* L.)] and plant development [PS: onion (*Allium cepa* L.), broad beans (*Vicia faba* L.), cucumber (*Cucumis sativus* L.), PE: duckweed (*Lemna minor* L.), PP: spring onion (*Allium fistulosum* L.)], ultimately resulting in a reduction in biomass. However, in some cases, there may be concentration-dependent negative and positive changes in photosynthetic indicators and chlorophyll content (Azeem et al., 2021). It is important to note that many of these experiments applied unrealistically high concentrations of plastics, making it challenging to draw definitive conclusions. Thus, there is a need for studies that use environmentally relevant plastic concentrations.

As demonstrated above, plastic particles in the micron range can disrupt plant growth and/or cause damage to oxidative antioxidant systems and even exert genotoxic effects. Additionally, smaller particles have been detected in edible species at worrying levels (up to 233 particles/kg), raising concerns about the presence of MPs (and particularly NPs) in these food sources and their potential transfer through consumption. Vegetables and fruits, for example, have been found to contain the highest abundance of MPs smaller than 10 μm, with fruits typically exhibiting the highest contamination levels. This may be attributed to the highly vascularised pulp, the extensive and complex root system of trees, and the advanced age of the plants compared to vegetables. All of these parameters can influence the extent of MP contamination (Campanale et al., 2022). In an agriculture-based economy, MPs can have detrimental effects, resulting in substantial economic and nutritional losses. Furthermore, the introduction and accumulation of MPs in the terrestrial food web can lead to unforeseen health consequences, highlighting the necessity for further research in this regard (Iqbal et al., 2023; Pérez-Reverón et al., 2022).

4.3.2. Microplastic effects on soil fauna

Soils serve as habitats for various animals (Wang et al., 2019), and the activities of soil fauna continuously shape these habitats, thereby performing vital functions within the soil ecosystem (Zhou et al., 2021a). They (a) contribute to the distribution of various substances through bioturbation, (b) participate in the decomposition of organic matter, and (c) assist in the control of soil-borne diseases and pests (Xu et al., 2020).

While considerable attention has been given to the health effects of MP pollution on aquatic animals, there is a lack of knowledge regarding the response of terrestrial fauna to this emerging pollutant. Furthermore, most studies have primarily focused on a few invertebrate taxa, such as oligochaeta (*e.g.* earthworms), nematodes, isopods, collembolan (*e.g.* springtails) (Zhu et al., 2019).

Based on their body size, soil fauna members can be categorised into three subcategories: microfauna (<0.1 mm), mesofauna (0.1–2 mm) and macrofauna (>2 mm) (Xu et al., 2020). Micro- and mesofauna occupy lower trophic levels in the terrestrial food web, and therefore serve as potential entry points for MPs into soil-dwelling animals and potentially humans. This can result in physical (*e.g.* skin lesions, digestive disorders) or biochemical damage (*e.g.* disturbances in carbohydrate and lipid metabolism or the osmotic system). However, it should be noted that some of these symptoms may not be directly caused by the ingestion of MPs but by secondary pollutants (Pérez-Reverón et al., 2022). In general, the effects of MPs on soil fauna become stronger with smaller particles sizes and higher abundance (Wang et al., 2019). For example, the uptake and accumulation of 0.1 μm PS-MPs (at a concentration of 1 mg/L) significantly reduced the survival rate, lifespan and body length of the nematode *Caenorhabditis elegans* and induced gene expression changes that led to irreversible damage to GABAergic and cholinergic neurons (Kumar et al., 2020). Moreover, MP ingestion in nematodes also caused reduced intestinal Ca^{2+} levels, oxidative stress and altered energy metabolism. These effects were more pronounced in species with faster life cycles and higher nutrient demands (Pérez-Reverón et al., 2022). Among the soil mesofauna, the springtail *Folsomia candida* serves as an ecologically relevant model organism. The growth and reproduction of this species were inhibited by 0.1% PVC-MPs, affecting its metabolism and gut microbiome composition (Kumar et al., 2020; Xu et al., 2020; He et al., 2018). Similar negative effects were observed with 0.1–1% PE-MPs (Zhu et al., 2019). Interestingly, *F. candida* tended to avoid soils with higher MP contamination, suggesting that springtails may be useful bioindicators of soil MP contamination (Xu et al., 2020; Zhu et al., 2019). As macrofaunal components, soil-dwelling worms are considered important ecotoxicology indicators of soil quality (Zhou et al., 2021a). While exposure to certain types of MPs may not necessarily result in reduced reproduction or survival rates among these organisms, other MPs can indeed exert toxicity. For example, PVC had minimal effects on the reproductive capacity of the annelid *Enchytraeus crypticus*, whereas nylon reduced it by 25% (Lahive et al., 2019). Similarly, HDPE had no significant effect on the mortality of the earthworm *Lumbricus terrestris* (Hodson et al., 2017), but LDPE did (Lwanga et al., 2016). High concentrations (28%, 45% and 60%) of PE-MPs significantly reduced the growth rate of *L. terrestris* and increased mortality (Lwanga et al., 2017). Even lower concentrations (1% and 2%) of PS-MPs showed similar effects on the earthworm *Eisenia fetida*, although these negative effects were less pronounced below 0.5% concentrations (Cao et al., 2017). These results suggest that MPs may have concentration-dependent and direct toxic effects on soil-dwelling organisms (Zhou et al., 2021a). Terrestrial worms play an important role in the movement of MPs in soil (Wang et al., 2019), as they can ingest and accumulate MPs and generate secondary NPs through plastic particle digestion. As a result of the activities of soil fauna members (*e.g.* ingestion, egestion, adhesion, burrowing, predation, *etc.*), MPs and NPs can be horizontally dispersed or distributed to deeper soil layers and transferred to other organisms, thereby increasing the bioavailability of both plastics and secondary contaminants such as heavy metals, persistent organic pollutants and antibiotics (Pérez-Reverón et al., 2022; He et al., 2018).

Similar to the phytotoxicity tests mentioned above, studies of ecotoxicological effects on soil fauna (a) often apply unrealistically high MP concentrations that may not be environmentally relevant, and (b) are mostly limited to the assessment of physiological and morphological responses. Remarkably little attention has been paid to changes in soil faunal biodiversity (Bouaicha et al., 2022).

It is important to consider that MPs can enter and accumulate in the food chain through the soil fauna. While the trophic transfer of MPs from phytoplankton to zooplankton and then finally to vertebrates in marine ecosystems is well-documented (Carbery et al., 2018), recent research has demonstrated a similar process occurring from soil to chickens *via* earthworms. The accumulation of MPs in the terrestrial food web is evident in the significantly higher levels of plastics found in earthworms (about 13 times more), chicken manure (almost 105 times more) and chicken gizzard (about 5 times more) compared to the soil itself (Banerjee and Shelver, 2021). In higher animals, MPs initially accumulate in the digestive tract and subsequently enter the circulatory system, muscles and other tissues. The presence of MPs in the body can lead to inflammation, developmental disorders and damage to the immune and reproductive systems (Qiu et al., 2022).

4.3.3. *Microplastic effects on soil microbiota*

In the soil environment, the rhizosphere plays a crucial role in shaping microbial diversity and activity (Bouaicha et al., 2022). The microorganisms inhabiting this region are key contributors to the soil ecosystem, since they influence soil fertility, crop yield and stress tolerance through their involvement in soil structure formation and biogeochemical cycles. These functions are vital for food production and climate protection (Hartmann and Six, 2023; Zhou et al., 2021a). Understanding the impact of MPs on soil microorganisms is therefore of outmost importance, as these changes can have far-reaching consequences at the ecosystem level (Zhou et al., 2021a).

4.3.3.1. Microbial community composition and potential functions. MPs can directly affect soil-dwelling microorganisms by providing novel habitats (Zhou et al., 2021a). For example, Mészáros et al. (2022) observed higher microbial cell counts in soil contaminated with fragments of disposable surgical mask compared to the uncontaminated control. This suggests that soil microbiota members were able to colonise the porous structure of the PP mask, but also raises concerns about the potential release of pathogens, adhering to the pores during mask use, into the environment later on. The microorganisms adhering to the soil/plastic interface form a unique community known as the plastisphere. There are notable differences in microbial community composition between the plastisphere and the surrounding habitat (Rillig et al., 2023), hypothesising that the presence of MPs may favour certain taxa and lead to the formation of microbial hotspots by disrupting belowground plants-microbe interactions (Zhou et al., 2021a). For instance, while the microbial diversity of the soil plastisphere can be generally lower than in the surroundings, certain bacterial taxa from the families *Vibrionaceae* and *Pseudoalteromonadaceae* are abundant on the surface of plastic debris but less so in their surrounding environment (Wang et al., 2019). Bacterial co-occurrence network analysis performed by Zhang et al. (2019b) revealed that the biotic interactions between microorganisms on MPs were as intricate as those observed in soil, with the members of the phyla Acidobacteria, Chloroflexi, Gemmatimonadetes and Bacteroidetes being keystone species. Zhang et al. (2023d) discovered that one-year exposure to various MPs at 2% favoured the enrichment of MP-adapted genera belonging to Actinobacteria (while the phyla Bacteroidetes and Gemmatimonadetes decreased). The soil plastisphere can also exhibit selective enrichment of microorganisms with genes responsible for antibiotic resistance or plastic biodegradation (Rillig et al., 2023). It was found that MPs promoted the enrichment of the potentially PE-degrading taxa, Actinobacteria, Bacteroidetes and Proteobacteria, acting as a “special microbial accumulator” in a farmland soil (Zhou et al., 2021a). This illustrates that niches created by MPs can contribute to the proliferation of certain microbial groups, which may have unpredictable consequences for ecosystem functions (Zhou et al., 2021a).

MPs indirectly impact the composition and functions of a microbial community by altering the environmental characteristics (*e.g.* soil aggregates and pore space, hydrological conditions, *etc.*) of the soil matrix, which serves as a habitat. For example, the creation of small drainage channels by MPs not only increases evaporation in the soil, but also enhances oxygen fluxes, leading to the proliferation of aerobic microorganisms (Qiu et al., 2022). Similarly, changes in soil porosity can affect the proportion of aerobic and anaerobic microorganisms. Additionally, the modification of soil nutrients by MPs can increase microbial activity and enrich species capable of biodegrading plastics and toxic compounds (Pérez-Reverón et al., 2022).

Although MPs have been shown to influence microbial community composition, metabolic rates and microbial functions (Wang et al., 2022b), the results are often contradictory. As demonstrated in Table 3, the presence of MPs in soils often leads to alterations in the abundance of bacteria belonging to the phyla Proteobacteria, Actinobacteria, Acidobacteria, Firmicutes and Chloroflexi, but clear trends are rarely observed. For example, the relative abundance of Proteobacteria increased at 0.0002–0.002% PE and PP (Yi et al., 2021), 1–20% poly (3-hydroxybutyrate-co-3-hydroxyvalerate) (PHBV) (Zhou et al., 2021c) and 2% LDPE (Rong et al., 2021), but decreased in the presence of 5% PE (Ren et al., 2020) - although in the latter case, changes were influenced by the size of the MP used. The relative abundance of Actinobacteria increased at concentrations of 0.01–1% PE and PLA (Zhang et al., 2023b); 0.1–10% PVC (Zhang et al., 2023c); 2% PS (Zhu et al., 2022a); 0.1–5% PP and PE (Yuan et al., 2023); 5% PE (Ren et al., 2020); 7–14% PS, PE and PVC (Fan et al., 2022); and 28% PE (Hou et al., 2021). In contrast, 1% PE and PP (Yu et al., 2021) and 2% PVC (Zhu et al., 2022a) showed an opposite effect, while 5% PE had no significant impact on Actinobacteria (Ma et al., 2023). Apparently, fungal and bacterial communities tended to exhibit distinct responses to the presence of MPs (Yuan et al., 2023; Fan et al., 2022; Hou et al., 2021). Furthermore, even bacterial taxa themselves demonstrated varying reactions depending on the polymer type and the MP dosage applied. In the experiments by Feng et al. (2022), most of the MPs [*i.e.* HDPE, PS, PLA, PBS and poly(3-hydroxybutyrate) (PHB)] dosed at a medium concentration (2%) increased the abundance of Actinobacteria, while PA increased the proportion of Proteobacteria. However, under the same conditions, Acidobacteria decreased. Rong et al. (2021) found that 2% LDPE increased the relative abundance of Proteobacteria but 7% LDPE increased Chloroflexi. In another study (Zhang et al., 2023c), 0.1% PVC also increased the relative abundance of Chloroflexi, but decreased at higher doses. Changes in the abundance of Proteobacteria are particularly interesting, since this taxon is often enriched in the soil plastisphere (Rillig et al., 2023).

Gao et al. (2021) found that the relative abundance of both Gram-positive and Gram-negative bacteria increased with increasing concentrations of LDPE-MPs (0.1–18%). On the other hand, in a field-scale experiment investigating the effect of LDPE-MPs, Brown et al. (2022) found no significant short-term variations in the soil microbial community composition. These results suggest that the scale of the experiment and the type of the polymer may be important factors in the effects of MPs on the microbial community. Similarly, the length of the experiment may be a determining factor in some cases. This was supported by the work of Ya et al. (2022), where at 1% PE supplementation, the proportion of Proteobacteria increased by day 7 of the incubation, while the relative abundance of Acidobacteria decreased by day 21 and that of Firmicutes by day 35. When the dose was increased (5% PE), similar changes occurred in these taxa, but this time the abundance of Actinobacteria also increased. Another example is the study by Xiao and colleagues (2022), where 0.01% and 1% PE increased the abundance of *Methanomassiliicoccus*, Acidobacteriales, Clostridia, Elusimicrobia, *Pseudomonadaceae*, *Deinococcaceae* and *Deinococci* by day 15, but the proportion of other taxa (*e.g. Reyranella*, *Alcanivorax*, Rhodospirillales, *Thermincola* and *Smithella*) did not increase until day 100 of the experiment.

The advent of high-throughput techniques, referred to as omics, enables the more detailed characterisation of microbial communities and metabolic pathways. For instance, next-generation sequencing (NGS) methods can be utilised to map not only the microbial diversity as discussed above, but also the potential functions of the microbial community (Laczi et al., 2020). These microbial functions play a crucial role in soil ecosystems as they can influence the mobility and cycling of certain elements, which in turn inevitably affects soil C and N pools (Bouaicha et al., 2022; Wang et al., 2022b). For example, during the 80-day experiment, the presence of 1% PS, PP and PVC enhanced amino acid biosynthesis and carbon metabolism, while PE and PP also enhanced functions related to environmental information processing and cellular processes (*e.g.* bacterial chemotaxis, flagellar assembly) (Yu et al., 2021). In soils co-contaminated with 5% PE and 20 mg/kg phenanthrene, Liu et al. (2022) observed enhanced metabolism of carbohydrates, amino acids and lipids (energy metabolism), as well as biosynthesis of secondary metabolites (bioconversion of toxic substances, protection against oxidative stress). Similarly, the relative abundance of genes encoding starch-degrading enzymes increased in the presence of 0.1–5% PP and PE (Yuan et al., 2023). In addition, Wang et al. (2023) detected a decrease in carbohydrate metabolism in the bulk soil contaminated with 0.5% PE, but an increase was observed in soil aggregates, whereas amino acid metabolism was overall increased. Using non-targeted metabolomics, Wu et al. (2022) found that the presence of LDPE not only increased carbohydrate metabolism but also the biodegradation of xenobiotics. Although the biosynthesis of cofactors and vitamins was reduced, it is possible that the soil studied has developed some resistance to secondary contaminants. Using NGS methods, the stimulation of genes regulating metabolic processes and biodegradation of xenobiotics has also been shown in PVC (Zhu et al., 2022a), PLA and PE-contaminated soils (Feng et al., 2022; Liu et al., 2022), further supporting this possibility. Similar findings were reported by Zhang et al. (2023d) after one-year exposure to various MPs (PE, PP, PA, PS, PET, PVC): microbial communities underwent changes that favoured xenobiotic and lipid metabolisms. Nevertheless, the increased oxidative stresses led to genetic damage in functions related to 'Replication and repair,' 'Folding, sorting and degradation,' and 'Transcription.' The alterations in microbial functional genes linked to C and N transformation, oxidation-reduction, and hydrolysis processes have potential implications for biogeochemical cycles.

From the above observations, it seems to be evident that the presence of MPs can stimulate C metabolism, but it can also alter functions related to N cycling. Qin et al. (2023) conducted an experiment in which different soil types were supplemented with 0.1% and 0.5% MPs (*i.e.* HDPE, PP and PS). The study revealed changes in N_2 fixation (*nifH*), N_2O reduction (*nosZ*) and denitrification (elevated *nirS* and reduced *nirK*) processes. These alterations resulted in increased levels of NH_4^+-N in farmland soil and decreased levels of NO_3^--N in all soil types studied. The findings highlighted the importance of soil type, as sandy soils, which generally have lower innate fertility than farmland and forest soils, were more susceptible to the negative effects of MP pollution (including the changes in physicochemical properties, as well as in bacterial community structure and functions). Similarly, 0.5% PVC supplementation increased the relative abundance of genes coding for urease (*ureA*, *ureB*, *ureC*, *URE*), nitrate reduction (*nasA*, *NR*, *NIT-6*) and N_2 fixation (*nifD*, *nifK*, *nifH*), but decreased the abundance of the nitrification gene *amoC* (Zhu et al., 2022b). In contrast, while urease-encoding (*ureA*) genes were upregulated, genes regulating N_2 fixation (*nifD* and *nifK*) and assimilated nitrate reduction (*narB*) were silenced in response to PE exposure (Yuan et al., 2023; Gao et al., 2021).

Table 3 demonstrates the use of a diverse range of soils and plastics in the experimental setups, and it is evident that neither the experimental

Table 3

The effects of microplastics (MPs) on soil microbial community composition and functions based in next-generation sequencing (NGS) data ('N. d.' and 'N. i.' denote parameters that are not detailed and not investigated with NGS, respectively).

Soil type or texture	Experiment		Polymer				Conclusion		Reference
	Scale	Duration	Type	Size	Shape	Concentration	Microbial community composition	Potential functions	
Farmland soil	200 g soil	35 days	LDPE, PE	1 mm	Sphere	1%, 5%	• 1% PE increased the relative abundance of *Proteobacteria* (by the 7th day), but decreased *Acidobacteriota* (by the 21st day) and *Firmicutes* (by the 35th day); • 5% PE increased the relative abundance of *Actinobacteria* (by the 7th day), but decreased *Acidobacteriota* (by the 21th day) and *Firmicutes* (by the 35th day); • MP supplementation enriched the genera of *Nocardia*, *Aeromicrobium*, *Amycolapsis* and *Rhodococcus*, while *Arthrobacter*, *Bacillus* and *Blastococcus* were predominant in response to PE-MPs	N. i.	Ya et al., 2022
Cropland soil (with sandy loam texture)	500 g soil	120 days	HDPE, PS, PA, PLA, PBS, PHB	39-80 µm	N. d.	0.2%, 2%	MP effects varied depending on their type and dosage: • 2% MP (except: PA) increased the relative abundance of *Actinobacteria*; • each type of MPs decreased the relative abundances of *Verrucomicrobia* (except: 0.2% PLA), *Armatimonadetes* and *Dependentiae*; • 2% PA increased the relative abundance of *Proteobacteria*; • 2% PA and bioMPs decreased the relative abundance of *Acidobacteria*	PLA and PE stimulated genes responsible for regulating xenobiotics biodegradation and metabolic processes	Feng et al., 2022
Agricultural soil (Eutric Cambisol with sandy clay loam texture)	Field scale	180 days	LDPE	N. d.	Powder	0.1-10%	No significant short-term changes in the microbial community composition	N. i.	Brown et al., 2022
Loamy and sandy soils	10 g soil	29 days	PE, PP	800 nm-3 mm	Film (PE), fibre (PP), microbead (PP)	0.0002%, 0.002%	The presence of MPs increased the relative abundances of *Deinococcus-Thermus* and *Chloroflexi*, while decreased *Proteobacteria, Acidobacteria, Bacteroidetes, Gemmatimonadetes*and *Firmicutes*	N. i.	Yi et al., 2021
Farmland soil	400 g soil	25 days	PHBV	N. d.	Pellet	1-20%	MPs increased the relative abundances of *Proteobacteria* and *Acidobacteria*, while decreased the relative abundance of *Firmicutes*	N. i.	Zhou et al., 2021c
Agricultural soil (Stagnic Anthrosol with loamy texture)	600 g soil	50 days	PE, LDPE, PVC	18-678 µm	Particle	1%, 5%	• PE pollution resulted in a greater reduction in operational taxonomic units (OTUs) • 5% PVC increased the relative abundances of *Betaproteobacteriales* and *Pseudomonadales*, while 1% PVC increased them; • 5% PVC decreased the relative abundance of *Sphingomonadaceae;* • 1% PE, 5% PE and 5% PVC increased the relative abundance of *Burkholderiaceae*, while decreased *Acidobacteria*	1% PE, 5% PE and 5% PVC improved functional genes assigned to membrane transport, while the relative abundances of functional genes assigned to metabolic processes, replication and repair were decreased	Fei et al., 2020
Farmland soil (with loam texture)	20 g soil	30 days	LDPE	80-450 µm	Particle	0,1%-18%	The relative abundances of Gram-positive és Gram-negative bacteria increased with MP concentration (highest at 18%)	18% LDPE caused a significant decrease in the abundances of ammonia oxidiser (*amoA*) and denitrification (*nirS*) genes	Gao et al., 2021

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Table 3 (*continued*)

Soil type or texture	Experiment		Polymer				Conclusion		Reference
	Scale	Duration	Type	Size	Shape	Concentration	Microbial community composition	Potential functions	
Field soil (with clay texture)	200 g soil	30 days	PE	<150 µm	Particle	5%	• The relative abundance of *Actinobacteria* increased until the phylum became predominant, replacing *Proteobacteria* • Plastic particle size influenced the effects of MPs on alpha diversity	N. i.	Ren et al., 2020
Agricultural soil	200 g soil	90 days	LDPE	150–250 µm	Powder	2%, 7%	• 2% LDPE increased the relative abundance of *Proteobacteria* and *Bacteroidetes*, while 7% LDPE increased *Chloroflexi*; • The genera *Pedomicrobium* and *Nocardia* were stimulated by LDPE-MPs	LDPE-MP treatment increased the abundance of *amoA*, *nirS*, *nirK* és *nifH* genes, while *nifH* and *nirS* decreased by the 90th day	Rong et al., 2021
Alluvial soil (with silty clay loam texture)	1 kg soil	40 days	PVC, PE, PS	10 µm	N. d.	10%	PVC and PE stimulated the abundance of Gram-positive and Gram-negative bacteria, while PS inhibited them	N. i.	Shah et al., 2023
Rice paddy field soil (Stagnic Anthrosol with clay texture)	60 g soil	100 days	PE	40–48 µm	Particle	0.01%, 1%	MPs increased the abundances of *Acidobacteriales* and *Pseudomonadaceae* (by the 15th day), as well as *Rhodospirillales*, *Reyranella* and *Alcanivorax* (by the 100th day)	N. i.	Xiao et al., 2022
Yellow-brown soil (with silty clay texture)	500 g soil	15 days	LDPE, PVC, PS	200 µm	Particle	2%	MP effects varied depending on their type: • PE stimulated *Patescibacteria* and *Bacteroidetes*, while inhibited *Proteobacteria*; • PS increased the relative abundances of *Proteobacteria*, *Actinobacteria* and *Gemmatimonadetes*; • PVC stimulated *Proteobacteria*, while inhibited the other taxa	• In contrast to PVC-MPs and PE-MPs, PS-MPs decreased functional category levels, including metabolism, cellular processing, genetic information processing and environmental information processing; • PVC improved such functional category levels as xenobiotics biodegradation and metabolism	Zhu et al., 2022a
Farmland soil	100 g	310 days	PE, PS, PVC	100 µm	Particle	7%, 14%	• Fungal and bacterial communities responded differently to MPs with PVC having the greatest impact; • MP treatments increased the relative abundances of *Proteobacteria* and *Actinobacteria*, decreased *Acidobacteria*, and had no effect on *Chloroflexi*	N. i.	Fan et al., 2022
Agricultural soil	N. d.	365 days	PE	<13 µm	N. d.	5%	MP treatment increased the relative abundances of *Sphingomonas, Gemmatimonas, Bacteria* and *Betaproteobacteria*, while decreased *Sphingobium*	MPs improved lipid, amino acid, carbohydrate and xenobiotics metabolisms, as well as secondary metabolite biosynthesis	Liu et al., 2022
Agricultural soil	72 g soil	150 days	PE	<100 µm	Particle	28%	• Fungal and bacterial communities responded differently to MPs; • PE increased the relative abundances of *Actinobacteria, Chloroflexi* and *Firmicutes* relatív abundanciája nőtt, while decreased *Gemmatimonadetes* and *Bacteroidetes*	N. i.	Hou et al., 2021
Farmland soil	2 L pots filled with soil + corn (*Zea mays*) was planted	30 days	PVC	15 µm	Powder	0.1-10%	• MPs stimulated *Actinobacteriota* and *Proteobacteria*, but inhibited *Acidobacteriota* and *Firmicutes*; • The relative abundance of *Chloroflexi* decreased at 0.1% PVC, but increased at higher MP concentrations	N. i.	Zhang et al., 2023c
Coastal saline-alkali soil	500 g	120 days	PP, PE	30-1000 mesh (approx. 595-13.5 µm)	N. d.	0.1-5%	• Fungi were more sensitive to MP treatments than bacteria with PE having a stronger effect; • MPs increased the relative abundances of *Actinobacteria, Proteobacteria* and *Bacteroidetes*	MP treatments increased the relative abundance of genes encoding starch degrading enzymes and urease (*ureA*), while the genes associated with nitrogen fixation (*nifD*, *nifK*) and assimilated nitrate reduction (*narB*) were downregulated	Yuan et al., 2023

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Table 3 (*continued*)

Soil type or texture	Experiment		Polymer				Conclusion		Reference
	Scale	Duration	Type	Size	Shape	Concentration	Microbial community composition	Potential functions	
Farmland, forest and sandy soils	100 g soil	60 days	HDPE, PP, PS	70 µm, 250 µm	Microbead	0.1%, 0.5%	• Bacterial communities in less fertile soils were more sensitive to the presence of MPs; • MPs increased the relative abundances of *Proteobacteria, Firmicutes, Actinobacteria* and *Sphingomonas*	MPs changed nitrogen fixation (*nifH*), N_2O reduction (*nosZ*) and denitrification (elevated *nirS* and reduced *nirK)* processes, resulting in increased NH_4^+ in farmland soil and decreased NO_3^- in all soil types	Qin et al., 2023
Agricultural soil	N. d.	90 days	PE	100 µm	Particle	0.5%	PE stimulated *Actinobacteria*, while inhibited *Proteobacteria, Chloroflexi, Acidobacteria* and *Cyanobacteria*	PE inhibited carbohydrate metabolism in the bulk soil, while stimulated amino acid metabolism in the aggregates	Wang et al., 2023
Agricultural soil (with sandy loam texture)	100 g soil	60 days	PVC	150-650 µm	Powder, film	0.5%	MPs increased the relative abundances of *Sinomonas, Amycolatopsis, Nocardia (Actinobacteria)* and *Bradyrhizobium, Burkholderia-Caballeronia-Paraburkholderia (Proteobacteria)*, while decreased *Conexibacter* and *Streptomycetaceae*	MP treatments increased the relative abundance of genes encoding urease (*ureA*, *ureB*, *ureC*, *URE*) and the genes associated with nitrogen fixation (*nifD*, *nifK*, *nifH*) or nitrate reduction (*nasA*, *NR*, *NIT-6*), while decreased the abundance of nitrification-associated genes (*amoC*) (as a result, NH_4^+ and NO_3^- were increased)	Zhu et al., 2022b
Black and loess soils (with clay and loam sand textures, respectively)	40 g soil	53 days	PE, PLA	N. d.	Film	0.5%, 1%	1% PLA-MPs exhibited a stronger effect on soil microbial community compared to PE-MPs, and increased the relative abundances of *Actinobacteriota, Gemmatimonadota, Myxococcota, Chloroflexi, Acidobacteriota* and *Bacteroidota*	N. i.	Shi et al., 2022
Paddy field soil	N. d.	30 days	PVC, PLA	155–180 µm	Particle	10%	• PLA stimulated *Xanthobacteraceae, Burkholderiaceae, Bacillaceae* and *Micrococcaceae*, but inhibited *Gaiellacea*; • PVC decreased the abundances of *Xanthomonadaceae, Bacillaceae* and *Burkholderiaceae*, while increased *Gaiellaceae* and *Sphinomonadaceae*	N. i.	Song et al., 2023
Agricultural soil (with silt loam texture)	25 g soil	100 days	LDPE	25–50 µm	Film	1%	LDPE increased the relative abundances of *Actinobacteria* and *Proteobacteria*, while decreased *Bacteroidetes*	N. i.	Dissanayake et al., 2022b
Agricultural soil (with loamy sand texture)	25 g soil	100 days	LDPE	100 µm	Film	0.1-7%	• MP treatments decreased the relative abundances of *Proteobacteria, Acidobacteria, Gemmatimonadetes* and *Verrucomicrobia*; • 0,1%, 3% and 7% LDPE stimulated *Chloroflexi, Firmicutes* and *Actinobacteria*, respectively	N. i.	Palansooriya et al., 2022
Purple soil	300 g soil	28 days	PE	300 µm, 600 µm	Particle	5%	• MPs induced changes in the relative abundances of *Proteobacteria, Actinobacteria* and *Acidobacteria*, stimulated *Nitrospirae*, but had no effect on *Chloroflexi*; • Potentially PE-degraders were stimulated	N. i.	Ma et al., 2023
Uncultivated soil	2 kg soil	300 days	PE, PLA	0.01 mm	Fragment	0.01-1%	MPs stimulated *Actinobacteriota, Firmicutes* and *Patescibacteria*, while inhibited *Chloroflexi, Acidobacteriota, Gemmatimonadota* and *Myxococcota*	N. i.	Zhang et al., 2023b
Agricultural soil	400 g soil	60 days	LDPE	1-10 mm	Film	0.2-2%	LDPE increased the relative abundance of *Actinobacteria, Bacteroidetes, Nitrospirae* és *Proteobacteria*, while decreased *Coriobacteriales* and *Enterobacteriales*	N.i. (but non-targeted metabolomics revealed that plastic film residues impacted the spectrum of microbial metabolites: carbohydrate metabolism and xenobiotics biodegradation was improved)	Wu et al., 2022

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Table 3 (*continued*)

Soil type or texture	Experiment		Polymer				Conclusion		Reference
	Scale	Duration	Type	Size	Shape	Concentration	Microbial community composition	Potential functions	
Wetland soil	200 g soil	30 days, 365 days	PE, PP, PA, PS, PET, PVC	N. d.	N. d.	2%	Long-term MP effects on soil bacteriome differed from the short-term effects: • MP-adapted bacteria belonging to *Actinobacteria* were enriched, while the phyla *Bacteroidetes* and *Gemmatimonadetes* decreased	• MP treatments enhanced microbial xenobiotic and lipid metabolisms, while increased oxidative stress inducing genetic damage; • Microbial functional genes associated with C and N transformation were shifted; • A rise of soil pathogens in response to MPs	Zhang et al., 2023d
Wetland soil	N. d.	80 days	PS, PVC, PP, PE	180-200 µm	N. d.	1%	• MP treatments increased the relatove abundances of *Proteobacteria* és *Firmicutes*, while decreased *Chloroflexi* (except: PVC); • PE stimulated *Bacteroidetes*; • PE and PP inhibited *Actinobacteria* and *Acidobacteria* (these taxa did not change or even increased in the presence of PS and PVC)	• MPs enhanced amino acid synthesis and carbon metabolism; • PE and PP enhanced functions related to environmental information processing and cellular processes (chemotaxis, flagellar assembly)	Yu et al., 2021

design nor the level of contamination was consistent. As discussed in Sections 4.1 and 4.2, MP effects can vary depending on differences in both soil properties (*e.g.* soil type, porosity, pH, SOM, soil moisture, *etc.*) and MP characteristics (*e.g.* polymer type, shape, size, concentration, *etc.*). Therefore, these factors provide an explanation for the inconsistency of results obtained in different soil ecotoxicity studies.

4.3.3.2. Microbial activity. Soil enzymes are protein molecules released into the soil solution by soil-dwelling microorganisms and plant roots. Functioning as catalysts, they promote various chemical reactions, facilitating the decomposition of organic matter and the mineralisation of nutrients, thus enabling microorganisms and plants to fulfil their energy and nutritional requirements. Soil enzymes play a pivotal role in the biogeochemical cycles of several elements, and as a result, changes in their activity can have a profound impact on ecosystem functions (Zuccarini et al., 2023; Qiu et al., 2022).

Microbial enzyme activity is typically higher in the rhizosphere compared to the bulk soil (Bouaicha et al., 2022). Soil extracellular enzymes, which contribute to around 40–60% of total soil enzyme activity, are stabilised in the soil matrix by forming complexes with humic colloids, clay or humus-clay complexes. As a result, their activity is generally independent of the number of living cells and often only shows a slight correlation with microbial biomass or soil respiration (with the intracellularly active dehydrogenase being an exception, as its activity is proportional to cellular activity) (Zuccarini et al., 2023; Alkorta et al., 2003). Since soil enzymes are highly sensitive to various environmental impacts, including soil stresses, they can be used as environmental biomarkers or qualitative microbial activity indicators (Wang et al., 2022b).

The classification of soil enzymes is based on the elemental cycles they participate in. For example, α- and β-glucosidases (which release glucose by cleaving starch and cellobiose, respectively), cellulose, invertase (which hydrolyses sucrose to glucose and fructose), dehydrogenase (which oxidises organic compounds), phenol oxidase and peroxidase (both of which oxidise lignin and humic compounds) are all involved in the decomposition of carbon compounds. Additionally, certain microorganisms can utilise MPs (mainly biodegradable ones) as carbon sources, leading to an increase in the activity of soil enzymes involved in C cycling (Zuccarini et al., 2023; Bouaicha et al., 2022). To estimate soil microbial activity, fluorescein diacetate (FDA) is commonly used as a substrate, as it can be hydrolysed by various enzymes such as non-specific proteases, lipases and esterases (Alkorta et al., 2003; Adam and Duncan, 2001). Thus, the overall activity of FDA hydrolases (FDAses) serves as an indicator of total soil microbial activity (Bouaicha et al., 2022; Joos and De Tender, 2022; Adam and Duncan, 2001). Other enzymes involved in soil nutrient cycling include chitinase (which hydrolyses chitin), leucine aminopeptidase (which hydrolyses leucine residues of amino acids), proteases (which degrade proteins) and urease (which converts urea to ammonia) for N release. Acid phosphatase and alkaline phosphatase are hydrolytic enzymes responsible for P mobilisation, since they cleave phosphate groups from esters of phosphoric acid (Zuccarini et al., 2023; Bouaicha et al., 2022). Although not directly related to nutrient cycling, soil catalase is a commonly used ecotoxicological parameter. Catalase is an intracellular enzyme that converts hydrogen peroxide to water and oxygen, indicating the presence of metabolically active aerobic microorganisms (Zhang et al., 2023b).

MPs have been shown to impact soil enzyme activity by influencing soil physicochemical properties and altering the composition of microbial community (Wang et al., 2022b). However, similar to the changes observed in microbial community structure, drawing definitive conclusions is challenging (Table 4). For example, an increase in FDAse activity was detected in agricultural soils at low concentrations (1% and 5%) of PE-MPs (Ya et al., 2022) and at relatively high concentrations (7% and 28%) of PP-MPs in loess soils (Liu et al., 2017). Similar effect was observed even when glyphosate (3.6 kg/ha and 7.2 kg/ha) (Liu et al., 2019) or phenanthrene (20 mg/kg) (Liu et al., 2022) were present as secondary contaminants alongside PP-MPs (7% and 28%) or PE-MPs (5%), respectively. In contrast, Oladele et al. (2023) reported a decrease in FDAse activities in 1–4% PS-contaminated agricultural soils at the end of the 30-day experiment, despite their values initially being higher than those in the control soil. Moreover, the activity of certain hydrolytic enzymes varied with the level of contamination: β-glucosidase increased with higher MP concentrations, while cellobiohydrolase was inhibited. In clay soils, even at very low concentrations (0.0002% and 0.002%) of PE and PP (Yi et al., 2021) or medium concentrations (1% and 5%) of LDPE, PE and PVC (Fei et al., 2020), a decrease in FDAse activity was reported. A similar effect was observed by Dissanayake et al. (2022) in loamy soil with 1% LDPE. Interestingly, in the latter two cases, the decreased FDAse activity did not result in decreased activity of other hydrolytic enzymes (*e.g.* urease, acid phosphatase). On the other hand, several cases showed an overall increase in soil enzyme activity in the presence of conventional MPs (Song et al., 2023; Fan et al., 2022; Ya et al., 2022; Liu et al., 2022, 2019; Zhu et al., 2022b. Lin et al., 2020; Huang et al., 2019) and biodegradable ones (Song et al., 2023; Schöpfer et al., 2022; Zhou et al., 2021c). This can be explained by an increase in soil porosity, which favours microbial colonisation, and by the fact that certain microbial strains can use MPs as a carbon source (Bouaicha et al., 2022). However, others have reported the opposite trend, a general decrease in enzyme activities, and hence soil health, when soils were contaminated with conventional MPs (Hou et al., 2021; Zhao et al., 2021; Yu et al., 2021, 2020). Nonetheless, it was also observed that the presence of MPs did not (or only very slightly) alter soil enzyme activity, whether conventional (Ma et al., 2023; Shah et al., 2023; Zang et al., 2020) or biodegradable MPs (Chu et al., 2023; Chen et al., 2020). Yet, in most cases, there is no clear trend, and both hydrolytic and oxidative enzyme activities can show considerable variability with MP contamination (Wang et al., 2023; Zhang et al., 2023b, 2023c; Feng et al., 2022; Pinto-Poblete et al., 2022; Xiao et al., 2022; Yi et al., 2021; Fei et al., 2020).

From the above, it is evident that soil MP pollution affects C, N and P cycling by altering the activity of soil enzymes. (The most diverse effects can be observed in the activity of enzymes involved in C and N cycling, consistent with the changes described in Section 4.3.3.1. regarding the expression of genes related to C and N metabolism.) What is particularly interesting, however, is that in some cases, the presence of MPs also affected the activity of oxidative enzymes such as phenol oxidase, laccase and peroxidase (Chu et al., 2023; Schöpfer et al., 2022; Hou et al., 2021; Yu et al., 2020). These enzymes are well-known for their role in lignin biodegradation, but their potential in neutralising xenobiotics (*e.g.* polyaromatic hydrocarbons) is increasingly being explored (Singh and Gupta, 2020; Noman et al., 2019). As MPs reduced the activity of these enzymes in most cases, it can be assumed that the resistance of the studied soils to certain secondary contaminants was also reduced.

A study by Fuller and Gautam (2016) revealed that, based on the available data, the MP contamination rate, even in industrial areas, is at most around 7%, suggesting that any research using lower MP concentrations could potentially produce environmentally relevant results. However, it is important to note that a significant number of the studies reviewed here have used elevated, and in some cases extremely high, MP concentrations (10–28%) (Shah et al., 2023; Song et al., 2023; Zhang et al., 2023c; Fan et al., 2022; Schöpfer et al., 2022; Hou et al., 2021; ZHou et al., 2021c; Yu et al., 2020; Zang et al., 2020; Liu et al., 2019), and even under these conditions, the results can be influenced by variations in polymer type and the experimental duration. Nevertheless, there are relatively few studies that have investigated the effects of MPs on soil over longer time scales (Zhang et al., 2023b, 2023d; Fan et al., 2022; Liu et al., 2022; Pinto-Poblete et al., 2022; Schöpfer et al., 2022; Hou et al., 2021; Lin et al., 2020. Yu et al., 2020), and our understanding of the long-term effects of MPs remains limited. In fact, conducting studies over longer periods can provide valuable information: the longer the monitoring period, the more accurate the long-term effects can be understood. Additionally, the ageing of MPs, whether natural or artificial, can further modify the observed effects, as illustrated in the phytotoxicity study by Lozano et al. (2023). This particularly important to consider in all ecotoxicity assessments because the majority of research is conducted using virgin plastics, which may not reflect the realistic scenario in contaminated areas, such as agroecosystems, where plastics are often already in a spent state and have undergone some degree of ageing before entering the soil. Therefore, it is encouraged to use naturally or artificially aged MPs in studies investigating the impacts of MP contamination in soil.

Enzyme activity can be influenced by various factors, including nutrient availability, temperature or soil physical and chemical properties (*e.g.* soil structure, water permeability, water retention, pH, *etc.*) (Zuccarini et al., 2023). Any factor that affects these parameters will inevitably impact soil enzyme activity as well. Similar to Table 3, the studies reviewed in Table 4 involve a wide range of soil types, plastics, concentration rates or experimental designs. Consequently, the variability of these properties might account for the conflicting or inconclusive outcomes observed across different soil ecotoxicity studies.

Soil respiration is largely dependent on soil microbial activity, with carbon dioxide being emitted from the soil primarily as a result of microbial decomposition processes (along with the contribution of plant root and soil faunal respiration). Since microbial activity is a major determinant of respiration, any changes in soil conditions (*e.g.* soil structure, porosity, moisture, pH) induced by the presence of MPs, as discussed above, can easily modify soil respiration. For example, alterations in soil hydrological conditions can lead to changes in respiration activity. On the other hand, microfibres can increase soil moisture, which, above a certain level, can inhibit microbial activity and consequently, reduce respiration (Wang et al., 2022b).

It is worth noting that soil microbial activity also plays a critical role in mediating the biogeochemical processes of greenhouse gas (GHG) emissions, including CO_2, CH_4 and N_2O. However, the presence of MPs is likely to influence these processes by altering soil properties and interacting with the soil microbiota (Wang et al., 2022b; Ren et al., 2020; Rong et al., 2021). Nevertheless, the extent of the changes induced by MPs on GHG emissions can be quite variable, depending on soil properties, MP characteristics and the duration of exposure. Conventional plastics, although rich in C, are relatively stable substances and therefore, less likely to undergo rapid mineralisation. In contrast, biodegradable plastics, which can be readily utilised as a carbon source by microorganisms, have the potential to contribute directly to CO_2 emissions. Considering the significant quantity of MPs present in the soil environment, it is essential that future research efforts focus on evaluating their importance in GHG emissions and their impact on global climate change (Wang et al., 2022b).

5. Conclusions

Over the past decades, plastics have emerged as a versatile and efficient raw material for human use. However, the widespread production and consumption of plastics have led to significant economic, health and environmental risks. Inadequate disposal of plastic waste, along with accidental or intentional littering have resulted in the pervasive presence of plastics in various environmental compartments worldwide. Moreover, under natural conditions, plastics can break down into smaller fragments known as MPs (<5 mm). Although the issue of MP pollution was initially associated with aquatic environments, the release of MPs into soils occurs on a much larger scale, posing a considerable burden on both natural terrestrial habitats and human-created environments like urban soils and agroecosystems. Soil is not only a natural component for wildlife but also plays a vital role in supporting human life as it directly impacts food production, which relies on maintaining soil health and fertility. Consequently, the accumulation of MPs in soil ecosystems has become a pressing global

Table 4
The effects of microplastics (MPs) on soil enzyme activity ('+' and '-' indicates increased activity and inhibition, respectively; 0 means no effect; while 'N. d.' marks parameters not detailed).

Soil type or texture	Experiment		Polymer				Tested parameter	Effect	Reference
	Scale	Duration	Type	Size	Shape	Concentration			
Farmland soil	200 g soil	35 days	LDPE, PE	1 mm	Sphere	1%, 5%	FDAse	+	Ya et al. (2022)
							Urease	+	
							Neutral phosphatase	-	
Cropland soil (with sandy loam texture)	500 g soil	120 days	HDPE, PS, PA, PLA, PBS, PHB	39–80 µm	N. d.	0.2%, 2%	Phosphatase	- (PLA: +)	Feng et al. (2022)
							Catalase	+	
Grassland soil (with loamy sandy texture)	20 g soil	31 days	PA, PC, PE, PES, PET, PP, PS, PUR	1.26–1.76 mm	Fibre, foam, film, fragment	0.4%	Acid phosphatase	-	Zhao et al. (2021)
							β-D-glucosidase	-	
							Cellobiosidase	-	
							N-acetyl-β-glucosaminidase	-	
Loamy and sandy soils	10 g soil	29 days	PE, PP	800 nm-3 mm	film (PE), fibre (PP), microbead (PP)	0.0002%, 0.002%	FDAse	-	Yi et al. (2021)
							Dehydrogenase	+/-	
							Phosphatase	+/-	
							Urease	+/-	
Farmland soil	400 g soil	25 days	PHBV	N. d.	Pellet	1–20%	β-glucosidase	+	Zhou et al. (2021c)
							Leucine aminopeptidase	+	
Grassland soil (with sandy loam texture)	3 kg soil	60 days	PES	1.28 mm	Fibre	0.4%	β-glucosaminidase	0	Lozano et al. (2021b)
							β-D-cellobiosidase	-	
							Phosphatase	+ (drought conditions)/- (well watered conditions)	
							β-glucosidase	+ (drought conditions)/- (well watered conditions)	
Agricultural soil	100 g soil	150 days	PE	100 µm	Film	28%	Catalase	-	Yu et al. (2020)
							Laccase	-	
							Manganese peroxidase	-	
							Urease	-	
							Glucosidase	-	
							Peroxidase	-	
Cinnamon soil	200 g soil	90 days	LDPE	2 mm	Fragment	0.0076%	Catalase	+	Huang et al. (2019)
							Urease	+	
							Invertase (sucrase)	+/-	
Agricultural soil (Stagnic Anthrosol with loamy texture)	600 g soil	50 days	PE, LDPE, PVC	18–678 µm	Particle	1%, 5%	Urease	+	Fei et al. (2020)
							Acid phosphatase	+	
							FDAse	-	
Alluvial soil (with silty clay loam texture)	1 kg soil	40 days	PVC, PE, PS	10 µm	N. d.	10%	Leucine aminopeptidase	0 (PVC)/- (PS, PE)	Shah et al. (2023)
							β-glucosidase	-	
							β-cellobiohydrolase	- (PS, PE)	
							Alkaline phosphatase	-	
							Urease	0 (PS)/+ (PE, PVC)	
Rice paddy field soil (Stagnic Anthrosol with clay texture)	60 g soil	100 days	PE	40–48 µm	Particle	0.01%, 1%	β-glucosidase	-	Xiao et al. (2022)
							Xylanase	-	
							Cellobiohydrolase	+	
							Chitinase	+	
							Leucine aminopeptidase	-	

(*continued on next page*)

Table 4 (*continued*)

Soil type or texture	Experiment		Polymer				Tested parameter	Effect	Reference
	Scale	Duration	Type	Size	Shape	Concentration			
Farmland soil	100 g soil	310 days	PE, PS, PVC	100 µm	Particle	7%, 14%	Catalase	+	Fan et al. (2022)
							Alkaline phosphatase	+	
							Urease	+	
Farmland soil (Acrisol with sandy loam texture)	200 g soil	30 days	PS	250–1000 µm	Fragment	0.5–4%	FDAse	+/-	Oladele et al. (2023)
							Leucine aminopeptidase	+/-	
							β-glucosidase	- (low MP concentration)/ + (high MP concentration)	
							Cellobiohydrolase	+ (low MP concentration)/- (high MP concentration)	
							Alkaline phosphatase	-	
Rice paddy field soil	1,5 kg soil	70 days	PLA	20–50 µm	Particle	2%	β-glucosidase	0/-	Chen et al. (2020)
							Urease	0/-	
							Catalase	0/-	
Agricultural soil (Luvisol with silty loam texture)	10 kg soil	510 days	PLA, PBAT	< 0.5 and 0.5–2 mm	Pellet	20%	Lipase	+	Schöpfer et al. (2022)
							β-xylosidase	+	
							β-glucosidase	+	
							N-acetyl-β-glucosaminidase	+	
							Phenol oxidase	+	
Grassland soil (Eutric Cambisol with silty clay loam texture)	500 g soil	30 days	PVC, PE	25 µm	N. d.	1–20%	β-glucosidase	-	Zang et al. (2020)
							Cellobiohydrolase	0	
							Xylosidase	-	
							Leucine aminopeptidase	0	
							Chitinase	0	
Agricultural soil	N. d.	365 days	PE	< 13 µm	N. d.	5%	Urease	+	Liu et al. (2022)
							FDAse	+	
							Dehydrogenase	+	
							Neutral phosphatase	0	
Agricultural soil	72 g soil	150 days	PE	< 100 µm	Particle	28%	Catalase	-	Hou et al. (2021)
							Urease	-	
							β-glucosidase	-	
							Manganese peroxidase	-	
							Lactase	-	
							Polyphenol oxidase	-	
Volcanic ash-derived soil (Andisol) mixed with sand	2 L pots filled with soil + strawberry (*Fragaria x ananassa* Duch) was planted	150 days	HDPE	2–5 mm	Particle	0.02%	Acid phosphatase	+	Pinto-Poblete et al. (2022)
							Urease	-	
							Dehydrogenase	-	
Farmland soil	2 L pots filled with soil + corn (*Zea mays*) was planted	30 days	PVC	15 µm	powder	0.1–10%	Catalase	+	Zhang et al. (2023c)
							Urease	+	
							Protease	+	
							Invertase (sucrase)	-	
							Alkaline phosphatase	-	
Cultivated loessial soil (Calcaric Cambisol with silty texture)	200 g soil	30 days	PP	< 180 µm	Powder	7%, 28%	FDAse	+	Liu et al. (2017)
							Polyphenol oxidase	+/0	
Huangmian soil (Calcaric Cambisol with silty texture)	200 g soil	30 days	PP	50–250 µm	Powder	7%, 28%	FDAse	+	Liu et al. (2019)
							Phenol oxidase	+	
Agricultural soil	N. d.	90 days	PE	100 µm	Particle	0.5%	Urease	+/-	Wang et al. (2023)

(*continued on next page*)

Table 4 (*continued*)

Soil type or texture	Experiment		Polymer				Tested parameter	Effect	Reference
	Scale	Duration	Type	Size	Shape	Concentration			
Agricultural soil (with sandy loam texture)	100 g soil	60 days	PVC	150–650 µm	Powder, film	0.5%	Urease	+	Zhu et al. (2022b)
Paddy field soil	N. d.	30 days	PVC, PLA	155–180 µm	Particle	10%	β-glucosidase	+	Song et al. (2023)
							Leucine aminopeptidase	+	
							Acid phosphatase	+	
Agricultural soil (with silt loam texture)	25 g soil	100 days	LDPE	25–50 µm	Film	1%	FDAse	-	Dissanayake et al. (2022b)
							Urease	+	
							Acid phosphatase	+	
Purple soil	300 g soil	28 days	PE	300 µm, 600 µm	Particle	5%	β-1,4-N-acetylglucosaminidase	+/-	Ma et al. (2023)
							β-1,4-glucosidase	0	
							Phosphatase	0	
Uncultivated soil	2 kg soil	300 days	PE, PLA	0.01 mm	Fragment	0.01–1%	Acid phosphatase	+	Zhang et al. (2023b)
							Catalase	-	
							Urease	-	
							Invertase (sucrase)	-	
							Nitrite reductase	-	
Chestnut soil (Haplic Kastanozem)	Field scale	30 days	PLA	90 µm	Fibre, powder	0.2%	β-1,4-N-acetylglucosaminidase	+	Chu et al. (2023)
							Leucine aminopeptidase	+	
							β-glucosidase	0	
							β-cellobiohydrolase	0	
							Alkaline phosphatase	0	
							Phenol oxidase	0	
							Peroxidase	0	
							β-xylosidase	+	
Wetland soil	N. d.	80 days	PS, PVC, PP, PE	180–200 µm	N. d.	1%	Invertase (sucrase)	-	Yu et al. (2021)
							Urease	-	
							Catalase	0/-	
Subtropical forest soil (with loamy sand texture)	Field scale	287 days	LDPE	37.13 µm	Fragment	11,400–39,172 fragments/kg	α-glucosidase	+	Lin et al. (2020)
							β-glucosidase	+	
							Cellobiohydrolase	+	
							β-xylosidase	+	
							Leucine aminopeptidase	+	
							Acid phosphatase	+	

concern, and understanding their impacts has become a critical task for environmental protection efforts.

In this review, we have provided a comprehensive summary of the effects of MP pollution on soil ecosystems, encompassing both natural and agroecosystems. Our focus was on exploring (a) the pathways through which plastics are transported into soil systems, (b) the changes that plastics undergo under environmental conditions, and (c) their interactions with soil components that lead to modifications in soil physicochemical and biological parameters. The data available from various studies indicate that while MPs can have some beneficial effects, such as promoting the proliferation of specific microbial genera and improving soil aeration, their overall impact on the soil environment is mostly negative. They can either directly or indirectly (*e.g.* by adsorbing toxic substances on their surface) affect the soil environment. MPs can enter the soil from multiple sources, with the most common being associated with agricultural practices like plastic mulching and the use of compost or sewage sludge. Consequently, soil samples from agroecosystems have been the most frequently used in risk assessment investigation, highlighting the significant role of agriculture in MP pollution. Furthermore, intrinsic traits of MPs, including polymer type, shape, size and abundance, have also been found to strongly influence their effects.

However, there remains a lack of sufficient information on the effects of MPs in soil ecosystems. Therefore, further research is necessary to bridge these knowledge gaps, preferably involving diverse soil types, polymer types and test organisms. Ecotoxicological research merits greater attention to monitor and comprehend the interactions between soil biota and MPs. Biomarker assays, such as enzyme activity measurements, can provide valuable insights into how individual organisms respond to the presence of MPs and how effectively they can cope with this emerging stressor in terrestrial ecosystems. Considering the magnitude of plastic pollution, a standardised system for categorising plastics and for testing methods is imperative to facilitate future assessment, management and comparison of the damage they cause.

6. Research gaps and future directions

Despite the growing body of research on the effects of MPs on soil ecosystems (Sun et al., 2022), there are still gaps in our understanding. Therefore, several aspects may be worth considering in future research:

- Compared to aquatic ecosystems, the effects of MPs on soils have only been studied for a relatively short time, and even less attention has been paid to agricultural soils, despite their vulnerability to MP contamination. As a result, the available knowledge is limited, and the assessment is further complicated by the fact that most of the current research is limited to soils from only a few countries.
- The lack of standardised experimental designs and methods for extracting and quantifying MPs from environmental samples can make it difficult to compare results, especially across different soil types, agricultural systems and ecoregions.
- There is relatively little research investigating the effects of MPs in soil microcosm systems or at field scale under realistic environmental conditions. Additionally, including as many different crops as possible in phytotoxicity experiments would be worthwhile, as it would help establish a threshold above which soil MP contamination can cause yield loss.
- To obtain a more accurate and realistic understanding of the effects of MPs, it is important to use realistic MP concentrations.
- Additives such heavy metals, pigments, phthalic acids esters, polybrominated diphenyl ethers and other biotoxic compounds can be released into the environment through degradation and ageing of plastics, subsequently leaching into soil moisture (Qin et al., 2021; Gunaalan et al., 2020). These leachates can lead to complex pollution, and their impact on soil biological activity is poorly understood. Therefore, there is an urgent need for research in this area.
- Since MPs can adsorb secondary contaminants, which may be released over time and pollute the soil, it becomes essential to study MPs of different properties (*e.g.* hydrophobicity, electrostatic properties, functional groups, *etc.*) and their adsorption processes, as well as the effect of combined pollution.
- To resolve the conflicting results, further experiments are needed to investigate the effects of MPs on pH, trace elements, as well as water and element cycling.
- Most of the experiments investigate MP-induced changes in soils over an average period of 30–90 days. However, this period is particularly short when considering how long they remain in the soil without decomposing, so they have a much longer duration of action. Moreover, their environmental impact can be significantly affected by plastic ageing processes.
- It would be essential to focus on the effects of plastics at the smallest scale possible: some NPs, for example, can pass through the plant cell wall or be absorbed into the cells of other organisms, where they may induce adverse physiological processes. Therefore, it may be useful to assess the risk of NPs to soil ecosystems and, through them, to human health.
- The development and application of microbial plastic degradation techniques (*e.g.* degradative enzymes) can offer promising solutions for efficient and eco-friendly remediation of MP-polluted ecosystems.

CRediT authorship contribution statement

Bodor Attila: Conceptualization, Software, Visualization, Writing – original draft, Writing – review & editing. **Feigl Gábor**: Conceptualization, Funding acquisition, Writing – review & editing. **Kolossa Bálint**: Conceptualization, Writing – original draft. **Mészáros Enikő**: Conceptualization, Writing – review & editing. **Kovács Etelka**: Conceptualization, Writing – review & editing. **Laczi Krisztián**: Writing – review & editing. **Perei Katalin**: Conceptualization, Writing – review & editing. **Rákhely Gábor**: Conceptualization, Funding acquisition, Writing – review & editing.

Declaration of Competing Interest

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

Data availability

No data was used for the research described in the article.

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