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REVIEW

Microplastics in agroecosystems: A review of effects on soil biota and key soil functions

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Abstract
Contamination of soils in agroecosystems with microplastics (MPs) is of increasing concern. The contamination of the environment/farmland soils with MPs (1 µm to 5 mm sized particles) and nanoplastics (NPs; <1 µm sized particles) is causing numerous effects on ecological soil functions and human health. MPs enter the soil via several sources, either from intentional plastic use (e.g., plastic mulch, plastic greenhouses, plastic-coated products) or indirectly from the input of sewage sludge, compost, or irrigation water that is contaminated with plastic. Once in the soil, plastic debris can have various impacts such as changes in soil functions and physicochemical properties and it affects soil organisms due to its toxic behavior. This review paper describes the different effects of plastic waste to understand the consequences for agricultural productivity. Furthermore, we identify knowledge gaps and highlight the required approaches, indicating future research directions on sources, transport, and fate of MPs in soils to improve our understanding of various unspecified abiotic and biotic impacts of MP pollution in agroecosystems.

KEYWORDS
agricultural soils, biodegradation, microplastics, soil biota, soil functions

1 | INTRODUCTION

The United Nations Environment Program has recognized plastic pollution as the most relevant terrestrial ecosystem pollution (UNEP, 2018). As ubiquitous materials of a modern lifestyle, plastics are used extensively worldwide in a variety of contexts. Around 4.9 billion metric tons are produced yearly, and ≈60% of all plastics are disposed of and are accumulating in landfills and the environment. However, if the management trends of waste will continue, it is predictable that ≈12 billion of plastic waste will end up in landfills or the terrestrial system by 2050 (Geyer et al., 2017). Plastic pollution in agricultural soils has received increasing attention because it is put in direct contact with plastic from intentional use (e.g., plastic mulch, plastic greenhouses, plastic-coated products) or from using plastic-contaminated supply of sewage sludge, compost, and irrigation water (Büks & Kaupenjohann, 2020; Manz et al., 2001; Weber & Opp, 2020).

A field study by Corradini et al. (2021) showed the principal role of sludge application in increasing farmland MPs contamination. The authors reported MPs contamination in sludge and soil between 18 and 41 particles g^{-1}, with an average of 34 particles g^{-1} in agricultural fields in Chile. Furthermore, as recently reported by Weber, Lügger et al. (2022), Weber, Hahn et al. (2022) and Weber, Opp et al. (2022) for soil monitoring sites in Hesse (Germany), the average concentration of plastic particles (p) per kilogram of dry soil matter at a soil depth of 30 cm was between 5 and 85 p kg^{-1} with a mean of 46.88 p kg^{-1} for particles of mean size 1.29 mm and size range 0.21–7.56 mm.

The increasing relevance of MPs in soil systems is also reflected by the fact that Lehmann et al. (2021) suggested that plastic screening should become a standard parameter when assessing soil health. It was estimated that 14% of the total plastics released into the environment goes into agricultural soils (Weber & Opp, 2020). Plastic residues, once released into the soil environment, will fragment into smaller sized particles often categorized by size as nanoplastics (NPs; <1 μm), microplastics (MPs; <5 mm), and macroplastics (MaPs; >5 mm) (Rillig et al., 2017), although there are alternative proposals of MPs size definition (e.g., mesoplastic: >5.0 mm, coarse MPs: 2–5 mm, and <2 mm MPs) (Weber & Opp, 2020). Furthermore, MPs are also categorized as primary and secondary categories of 1–5000 μm in size. Primary MPs are purposefully created in smaller sizes, and secondary MPs are degraded from larger plastic particles (O'Brien et al., 2021).

Despite the increasing global awareness of MPs and NPs contamination in farmlands, the current conceptual understanding of the behavior and transformation of plastics in soil, especially with the combination of different sizes and concentrations of MPs, is insufficient to conclude their potentially harmful effects on soil functions, soil physicochemical properties, microbial communities, and plant growth (Wang et al., 2020). Although there are many recent review papers published about plastic pollution in farmlands and its impacts on soil, those papers never considered the combined effects (abiotic and biotic consequences of adding MPs at various sizes and concentrations) of MPs, revealing obvious knowledge gaps.

The present review summarizes the current status of knowledge on potential effects of MPs pollution on agricultural soils by reviewing the possible sources, transport, and fate of MPs in the soil environment. The aims are to (1) analyze and compare MPs influences on soil physical, chemical, and physicochemical functions; (2) summarize the impacts of MPs on soil biota; and (3) discuss the MPs impacts on plant growth parameters, entry to soil and ground water and the food chain, including the aspect of distribution, absorption, and toxicological effects on soil–plant systems.

2 | REVIEW SCOPE AND APPROACH

In the current review paper, materials such as statistical bulletins, journal articles, and conference papers were used for sourcing information. The following keywords were selected: "microplastics," "plastics" and "agricultural soil," "biodegradation," "groundwater," "fragmentation," "soil microbiome," "physical properties," "ecological risk," "sources," "chemical properties," "transport," "fate," "physicochemical properties," "plant growth parameters," "detection methods," and "food chain" to retrieve the publications from the Web of Science, PubMed, SCOPUS, Research Gate, and Google scholar from its inception in 1991 (the earliest time of presence of MPs in soil in scientific reports) to 2022. For the review preparation, six sets of literature data were collected: (1) MPs sources and distribution, (2) MPs impact on soil physical–physicochemical, (3) chemical and (4) biological characteristics, (5) plant growth–food chain, and (6) transport–groundwater. We found 186 articles of both research papers and review papers that were divided into six topics: (1) MPs sources and distribution (27 studies), (2) physical–physicochemical (31 studies), (3) chemical (20 studies), (4) biological (49 studies), and (5 and 6) plant growth–food chain and transport–groundwater (59 studies). The diagram of article proportion in each of the involved topics in this review paper is available in Figure S1.

3 | SOURCES AND DISTRIBUTION OF MPS IN AGRICULTURAL SOILS

The sources of MPs entering the soil can be divided into primary and secondary plastic residues, categorized based on their original manufactured size (R. Qi, Jones et al., 2020). Primary MPs are mainly produced in minuscule sizes (e.g., preproduction pellets, microbeads, and industrial materials such as cosmetics and detergents). In contrast, secondary MPs may originate from breaking down of larger plastic products due to solar UV radiation and soil biotic and abiotic processes (Okoffo et al., 2021).

Distribution and movement of MPs in soil environments occurs through multiple ways (Figure 1). In farmlands, entry is mainly due to fragmentation of larger plastics (e.g., from plastic mulch, greenhouse covers, poly-tunnels, silage baling, containers, packaging, and netting; Van Schothorst et al., 2021), atmospheric deposition of airborne MPs (e.g., plastics from uncovered landfills and urban littering; Allen et al., 2019), irrigation with contaminated water, plastic-coated seeds/pesticides/fertilizers, application of biosolids (e.g., compost and sewage sludge [He et al., 2020; Okoffo et al., 2019; Weber & Opp, 2020]), flooding (Rolf et al., 2022), organic farming with biodegradable plastics (Serrano-Ruiz et al., 2021), street runoff (e.g., tire and road wear particles; Sommer et al., 2018), and soil biota and animals' activities (Rillig et al., 2017; Yu et al., 2019).

Transportation of plastic residues also depends on MPs properties such as size, composition, hydrophobicity, surface charge, density, and shape (Lehman et al., 2021). A field study by Weber, Opp et al. (2022) in floodplain soils of the Lahn River (Germany) showed the vertical transportation of MPs in different size fractions of 500–2000 μm down to a depth of 2 m through bioturbation and preferential flow (Weber, Opp et al., 2022).

As a result, agricultural soils are polluted with primary MPs (e.g., released fibers from synthetic textiles, preproducts of manufacturing plastic products, microgranules added to detergents as foam suppressants, personal care products, medical and hygiene products) and

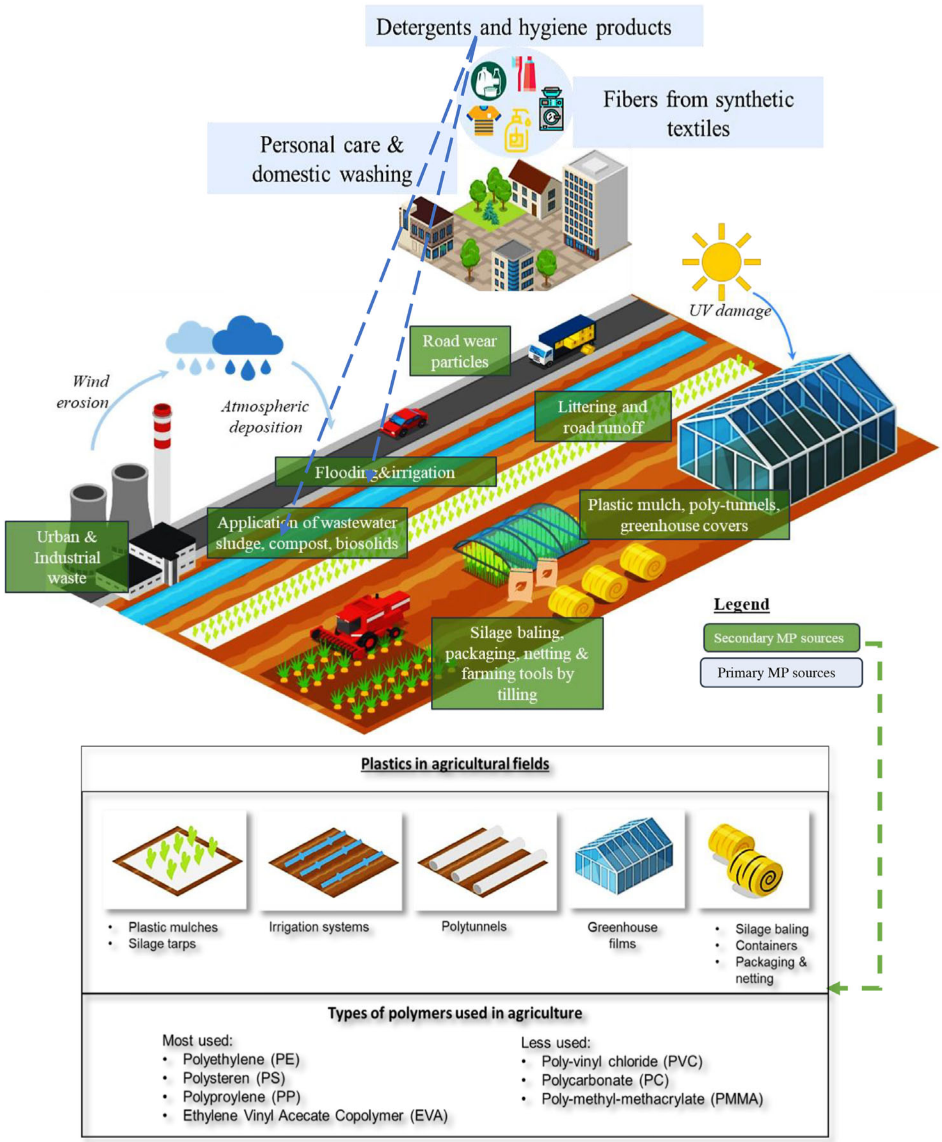

FIGURE 1 Possible input sources of microplastics into the agroecosystem (top) and use and polymer type of plastics in agriculture (bottom)

secondary MPs (e.g., application of plastic products such as mulches, greenhouse coverage, compost, and sludge; Anik et al., 2021; He et al., 2020). Primary MPs may enter the agroecosystem due to poor waste management, such as improper plastic removal and storage or on-site burning (Hopewell et al., 2009), which mainly happens near urban regions. Secondary MPs, on the other hand, are the prominent sources of plastic residues in farmlands due to the vast application of plastic in croplands and pastures (Büks & Kaupenjohann, 2020; Corradini et al., 2021; Ding et al., 2020) (Figure 1). Brandes et al. (2021) have studied the spatial modelling of farmlands MPs, which indicated that agricultural practices such as cropping system (e.g., greenhouse crops, or particular crops under plastic coverage) and farming management (e.g., fertilizers, organic amendments, irrigation, harvesting, and storage) play a crucial role in MPs distribution into the soil environment. Consequently, in the soil system, plastic may alter interactions between particles, water, chemicals, and microbes (Horton et al., 2017; Nizzetto et al., 2016) and, therefore, the accumulation of MPs in cultivated lands may adversely affect the various properties of agroecosystems.

4 | ABIOTIC IMPACTS OF MPs ON SOIL PROPERTIES

4.1 | Effects of MPs on soil chemical properties

The presence of MPs in soil has been reported to change soil chemical properties such as soil organic matter (SOM) content, pH, electrical conductivity (EC), organic carbon storage (by impact on, e.g., soil microbial processes or litter decomposition; Lian et al., 2021; Rillig et al., 2021). However, the effect varies based on the extent of exposure, quantity, type, and size of the MPs. Boots et al. (2019) found that soil pH, after 30 days of exposure to fibers added at 0.001% (w/w), high-density polyethylene (HDPE) and biodegradable polylactic acid (PLA) both added at 0.1% (w/w), was reduced by up to 0.6 pH-units compared with the control. In contrast, Y. Qi et al. (2019) found in a field study that the presence of MPs (low-density polypropylene [PP]) and biodegradable materials, added at 1% (w/w), caused an increase in soil pH after 2 months of exposure compared with the control treatment, while after 4 months of exposure, soil pH was decreasing again (Qi et al., 2019), pointing out the exposure time is a controlling factor on pH-changes by MPs. Soil pH may increase due to soil aeration and porosity, leaching the MPs' chemical additives to the soil and converting organic N to inorganic NH_4^+ (Zhao et al., 2021). However, shape, type, various MPs additives, soil biota enzymatic activities, and plant species have prominent roles in altering soil pH affected by MPs (Lozano et al., 2021). An increase in soil EC was reported in a field study of polymer-coated fertilizers effects on maize growth by Lian et al. (2021). Y. Qi et al. (2019) also reported the highest EC exposed to low-density polyethylene (LDPE) (390 ± 119.39 μS cm^{-1} at 2 months, 179 ± 76.73 μS cm^{-1} at 4 months) and lowest EC for biodegradable plastics (130 ± 48.42 μS cm^{-1} at 2 months, 75 ± 15.58 μS cm^{-1} at 4 months) where the initial EC was 411 ± 18.33 μS cm^{-1}. Furthermore, MPs can potentially impact the soil C:N ratio. For example, Y. Qi et al. (2019) mentioned that both LDPE and biodegradable plastics caused a significantly higher soil C:N ratio than the control after 2 and 4 months, whereas in a laboratory study by Zhong et al. (2021), with addition of PE particles (concentrations: 21.36%: <100 μm; 68.43%: 100–200 μm; 10.21%: >200 μm), soil C:N ratio decreased in presence of earthworms. Earthworms can potentially lower the C:N ratio by adding mucus, body fluids, enzymes, excreta as sources of N to the soil substrate (Arora et al., 2020; Zhong et al., 2021).

The concentration of SOM is also affected by MPs size, type, and concentration. A field study by Dong et al. (2015) showed a reduction in SOM, by applying large sizes of plastic mulch film (0–200 cm^2) and concentrations ranging from 250 to 2000 kg m^{-2} in cotton fields. Boots et al. (2019) showed that the presence of clothing fibers increased SOM, whereas HDPE and biodegradable PLA reduced SOM in comparison with the control. The plastic types were found to significantly affect the SOM with positive and negative priming effects. The positive effect occurs due to lower persistence and easier degradation of some plastic types (e.g., biodegradable plastics) that enhance the C source, microbial activity and growth, and exoenzyme activity, potentially leading to the enhanced mineralization of native SOM by cometabolism (Zhou et al., 2020). According to the organo–organo persistence hypothesis, the negative effect arises from diluting and adsorption of soil available C (dissolved organic carbon) to plastic surfaces (Rillig et al., 2021). However, the combined effects of MPs size, shape, type, and concentrations on soil organic carbon dynamics still remain unclear. Hence, studies using long-term field experiments with various combinations of size and concentrations of MPs in agricultural soil at different latitudes and climates to monitor MPs impacts on dissolved organic matter and specifically on soil organic carbon and nitrogen dynamics are needed. Similarly, studies should be conducted in mesocosm experiments under controlled and defined laboratory conditions.

4.1.1 | MPs additives and sorbed contaminants

MP additives can be categorized into colorants, functional additives, fillers, and reinforcements (Bridson et al., 2021). Most additives are hazardous (e.g., phthalates, chlorinated paraffin, and brominated flame retardants). In some instances, additives are not chemically bound to the polymer matrix and have the potential to leach from MPs and additionally to MPs contaminate the environment (Groh et al., 2019). Plastic producers use a diversity of polymers combined with different additives to obtain plastics of specific properties. Plastic additives facilitate the manufacturing and adjust the plastic properties such as color, strength, heat resistance and elasticity (Steinmetz et al., 2016). The degradation of plastic debris can release additives or their metabolites with various potentially toxic effects. Indeed, many plastic additives are suspected to be endocrine disruptors, these substances may interfere with animal hormones and impact the entire organism (Hermabessiere et al., 2017). Hazardous additive leachates can greatly affect plants and soil biota in a terrestrial soil environment. Although qualitative and quantitative detection approaches are still required to assess the MPs risk (Bridson et al., 2021).

Additionally, depending on the different sources, plastic debris may sorb various pollutants such as pharmaceuticals in wastewater, pesticides in agricultural fields, or antibiotics in manure (Wang, Liu et al., 2019). The sorption, release rate, and quantity of pollutants vary between the contaminant type, age of plastic, and the matrix' properties (water, soil, animal tissue). For example, pesticides were shown to adsorb stronger on polyethylene (PE) mulch film (584–2284 μg pesticide g^{-1} plastic) than on soil particles (13–32 μg pesticide g^{-1} soil) (Ramos et al., 2015). The alteration in MPs physicochemical properties due to the aging process (e.g., UV irradiation, thermal and biological degradation) can trigger different chemical effects of aged MPs in contrast to fresh plastics in the environment (Kublik et al., 2022). MPs can be distributed in the soil environment for a long time and exposed to UV irradiation, oxidation reactions, thermal degradation, and biodegradation. These environmental processes can modify MPs properties, morphology, mechanical strength, and amount of oxygen-containing functional groups (Liu et al., 2019). Furthermore, a recent analytical laboratory study by Lan et al. (2021) revealed a higher adsorption capacity of hydrophobic pesticides (e.g., carbendazim, diflubenzuron, malathion, and difenoconazole) on aged PE MPs compared with fresh ones in agricultural soils. An essential aspect of predicting the sorption rate is the hydrophobicity of the plastic and the contaminant. Some persistent organic pollutants (POPs) have a higher affinity to certain MPs due to the hydrophobic nature of plastic polymers. For example, Bakir et al. (2012), in their sorption behavior model study, showed that the plastic type has a decisive role in organic pollutants accumulation. The results confirmed that POPs were adsorbed onto plastic debris. There is evidence that particular POPs exhibit preferential sorption onto plastic polymers (e.g., PE and PP) compared with their sorption potential onto natural sediments (Teuten et al., 2009).

When plastic debris is ingested, additives and sorbed contaminants may leach into the organism's digestive system and bloodstream. For example, MPs were shown to act as carriers for pollutants in a study with lugworm *A. marina* (Teuten et al., 2009). In another laboratory study, Hodson et al. (2017) measured the bioaccumulation of zinc (Zn) contamination in *L. terrestris* from soil to gut in the presence of HDPE. Their gut model showed higher desorption rates from the MPs than from the soil, suggesting that HDPE may increase Zn bioavailability. However, no increased mortality or weight change of the earthworms was observed. Still, MPs could act as vectors to enhance metal exposure, such as for Zn. The metal exposure may increase by decreasing plastic sizes, resulting in a higher surface area and higher uptake of Zn by soil organisms like earthworms (Hodson et al., 2017). A field study by Weber, Hahn et al. (2022) investigated the spatial distribution of heavy metals and MPs in floodplain soils at the Lahn River, located in central Germany. The study's results revealed the spatial correlation of similar transport, release, and deposition pathways of those two contaminants (MPs and heavy metals) in the upper part of the soils (0–50 cm) in the floodplain system.

4.2 | Effects of MPs on soil physical properties

4.2.1 | Soil structure and aggregate stability

Soil structure plays a crucial role for edaphic conditions, soil fertility, soil water dynamics, and air permeability, all supporting plant and animal life. The structure of soil is associated with aggregate stability, soil water movement and retention, erosion, crusting, nutrient cycling, root penetration, and crop yield (Bronick & Lal, 2005; Uteau et al., 2013). Soil aggregate stability may be affected by MPs via various mechanisms depending on the bonding agents' nature (hydrophobic vs. hydrophilic substances; Bronick & Lal, 2005). For example, a field study by Vogelmann et al. (2013) showed a positive correlation between hydrophobicity and aggregate stability, which indicated that hydrophobic substances cover aggregate surfaces and act as cementing agents, thereby increasing aggregate stability (Vogelmann et al., 2013). In contrast, MPs are recognized as a material with hydrophobic surfaces (Kwon et al., 2017), decreasing soil aggregate stability (de Souza Machado et al., 2018). de Souza Machado et al. (2019) found a decrease in water-stable aggregates in various MPs-treatments, which was attributed to the MPs type (e.g., PET, PS, PP). Concomitant, more water-stable aggregates were observed in the rhizosphere of treated soils with MPs and plants (de Souza Machado et al., 2019). However, research on the effects of various MPs types and shapes on aggregate stability is still required.

MP inclusion into aggregates can enhance their accumulation in soil due to reducing mobility. The plastic shape also matters in that process. Comparing the effect of fibers, films, foams, and fragments with different chemistry, Lehmann et al. (2021) found that fibers negatively affected aggregation, irrespective of MPs chemistry. For other shapes, the chemical composition of MPs determined their effect on soil aggregation. A field study by G. S. Zhang and Liu (2018) showed that about 72% of total used plastic particles in the experiment were incorporated into soil aggregates, and 28% of plastic particles were dispersed within the experimental soil areas. The primary form of that plastic debris in soil was fibrous, making up 92% of each sample, followed by fragments and films that contributed to 8%. The abundance of aggregate-associated plastic fibers is higher in microaggregate than in macroaggregate fractions. In contrast, lower concentrations of plastic films and fragments were found in microaggregates (GZhang & Liu, 2018).

4.2.2 | Water retention and infiltration capacity

Consistent with MPs effects on soil structure, alteration in total soil pore space and hydrological parameters (e.g., evapotranspiration, pore structure, water-holding capacity, and hydraulic conductivity) was observed in studies by de Souza Machado et al. (2019, 2020). The authors reported that water-holding capacity was increased by adding polyester fibers in loamy sand soil, whereas PE fragments showed a low impact on water-holding capacity. In contrast, G. S. Zhang et al. (2019) reported a reduction in water-holding capacity after adding polyester fibers to clayey soil because of a significant decrease in the amount of <30 μm pores due to clogging by the fibers and increasing soil water repellency.

In a laboratory study by Wan et al. (2019), increasing MPs concentration reduced soil structural integrity and increased soil water evaporation. The authors suggested that plastic contamination could alter the water cycle in soils, which may exacerbate soil water shortages and drainage fluxes.

4.3 | Effects of MPs on soil physicochemical properties

Another expected effect of MPs on soil is the alteration in soil wettability, which is already addressed in Section 4.2.1. Changes depend on chemical (e.g., hydrophobic surfaces of MPs) and physical features (e.g., size and shape of MPs; Cramer et al., 2020). For example, a laboratory study by Y. Qi et al. (2020) showed a higher water repellency of soil treated with two MPs sizes (macro size: 6.92 ± 1.47 mm and micro size: 50–500 μm) and various concentrations (0.5, 1, and 2%) of MPs derived from plastic mulch residues. They observed that use of micro size (e.g., LDPE) resulted in longer water drop penetration times (WDPT) than macro size plastics, WDPT depended on concentration. Furthermore, the authors reported significant differences between types of MPs, with greater WDPT for biodegradable plastic compared with LDPE.

Cramer et al. (2020) showed an increasing contact angle with increasing MPs concentration in a laboratory experiment. The hydrophobic surface of MPs and their action as a hydraulic barrier depend on the MPs particle size and could potentially affect soil hydrological parameters (Y. Qi et al., 2020). This implies that MPs in farmlands could negatively affect plant growth parameters. A recent laboratory study examined the effects of MPs properties (e.g., shape, type, and surface chemistry) on soil geochemical and biophysical characteristics, which showed that MPs potentially could exacerbate adverse effects of other environmental stressors like droughts (Lozano et al., 2021). In general, most published papers on MPs contamination of farmland soils focus on chemical and biological problems caused by MPs, whereas studies on soil physical parameters (e.g., void ratio, shear strength, and hydraulic conductivity) are still scarce.

5 | EFFECTS OF MPs ON SOIL BIOTA

One of the essential parameters of soil health is the activity of soil microbes, which catalyze many of the biogeochemical transformations that determine soil quality and fertility, and plant growth, therefore providing food security to humans (Hu et al., 2020; Trap et al., 2016). The complex microbial network composed of bacteria, archaea, fungi, protozoa, and algae plays a critical role in agroecosystems (Richardson, 2001; Toor et al., 2020). Imbalances and alterations in the microbial community functioning and structure can have dire consequences for the whole system and, therefore, on crops in agricultural fields (Shi et al., 2022; Tripathi et al., 2020; Zaller et al., 2018). Furthermore, plants are interconnected with the soil's microbiome and fauna. They depend on this network for various functions such as growth and development, protection against pathogens, productivity and yield, and nutrient mobilization (Rillig et al., 2019). Only few studies are available addressing the effects of emerging ecosystem stressors such as MPs, on soil microorganisms in terrestrial ecosystems. However, recent studies reported that MPs physicochemical properties, such as particle size and polymer density, strongly impact soil microorganism's activities through an effect on crop metabolomics (e.g., changes in amino acids, saccharides, and organic acids) and thus decreasing crop biomass by about 25.9% (Wu, Liu et al., 2020). Microbial activity significantly decreased due to phthalate acid esters (PAEs) released from the plastic residue, depending on the plastic type and volume (Wang et al., 2016). Zhao et al. (2021) indicated that MPs could decrease microbial activity, depending on shape, polymer type, and exposure time. Moreover, the effects of MPs on different plants and microorganisms are also linked to MPs concentrations, shapes, and sizes (Kim & Rillig, 2022).

5.1 | Bacteria

The effects of MPs on bacterial community composition and structure are still poorly understood. In a study conducted in cotton fields in China, Zhang et al. (2019) noticed that MPs might act as "special microbial accumulators" where bacterial communities colonize polymers, while these communities have a significantly different structure from the ones in the surrounding soil. In a similar study, Puglisi et al. (2019) found that distinct bacterial communities were associated with different types of plastic and that the structure of the plastisphere may be correlated with physio-chemical properties of the polymer. Bacterial communities are a fundamental functional part of many organisms' organs, such as their gut. It was evaluated that even at low concentrations of 0.5% dry weight of soil, MPs markedly changed the microbial community and decreased bacterial diversity in the gut of springtail microarthropods (Ju et al., 2019). In a laboratory study by Zhu, Fang et al. (2018), gut bacterial communities were altered in *F. candida* exposed for 56 days to MPs, with enhanced bacterial diversity in treated organisms.

5.2 | Fungi

Studies on the direct adverse effects of MPs on mycorrhizal fungi are scarce. Nomura et al. (2016) conducted experiments on model fungi species and reported that NPs might be toxic to some fungi, although uptake depended on cell wall consistency. The detrimental impact may be linked to the charge of the particles and the nutrient solution utilized (G. S. Zhang et al., 2019). It can be hypothesized that *arbuscular mycorrhizal fungi* (AMF) may also be indirectly affected due to the effects of MPs on soil properties like bulk density, pore structure, water retention, and soil aggregation that condition AMF (Leifheit et al., 2021). Indeed, some plastic mulches indirectly impact fungal and bacterial communities by increasing or decreasing those microorganisms' abundance through the availability of nutrient (e.g., exchangeable potassium and nitrate; Frey et al., 2008; Koitabashi et al., 2012; Muñoz et al., 2017; Shan et al., 2022).

5.3 | Protists

Protists support essential ecological functions and stimulate plant growth-promoting rhizobacteria (Trap et al., 2016). Only a few protist species have been reported to uptake plastic microbeads. Boenigk and Arndt (2002) reported that *Acanthamoeba* and heterotrophic nanoflagellates ingested latex beads. In another laboratory study, the population abundance, biomass, and volume of free-living marine ciliated protozoa *Uronema marinum* were reported to significantly decrease after exposure to polystyrene (PS) beads (Y. Zhang et al., 2021). However, further studies are needed on MPs' potential effects on soil protozoa as they play a fundamental role for soil fertility. Furthermore, the activity of soil protozoa mineralizes organic soil nitrogen and phosphorous into ammonium and orthophosphate forms, respectively, thus enhancing plant growth (Chitra, 2017).

5.4 | Invertebrates

The ingestion of MPs and NPs from organisms at different terrestrial trophic levels has shown varying effects on growth, reproduction, fitness, and tissue damage. It should be noted that particle size affects organisms, such as earthworms, small rodents, and mammals, causing intestinal blockages (Huerta Lwanga et al., 2016, 2017). Smaller particles (<1 mm) are easier to be ingested. They could travel down to deeper soil depths and enter the food chain, causing internal tissue damage and risk disruptions at the endocrine level (Rodriguez-Seijo et al., 2017; D. Zhu, Chen et al., 2018). Effects of polyester fibers at various sizes and concentrations were tested in a laboratory study by

Selonen et al. (2020). Results showed evidence for smaller size fibers uptake in enchytraeidae and isopods. The uptake of smaller size MPs by organisms poses a risk of MPs entering the food web and hence has the potential to become a long-term threat for invertebrates and their predators (Selonen et al., 2020; B. K. Zhu, Fang et al., 2018).

5.5 | Collembola

Collembola species *F. candida* and *P. minuta* influence the translocation of MPs with dependence on particles and organism size (Maaß et al., 2017). Collembola's growth and reproduction were also inhibited (B. K. Zhu, Fang et al., 2018). Similar results were found by Ju et al. (2019), where *F. candida*, exposed for 28 d to PE (≤500 μm), showed avoidance behavior together with a 70.2% decrease in reproduction at 1% (w/w) soil dry weight concentration, and significantly altered microbial gut community. In general, by considering the exposure of soil microorganism species to higher concentrations of MPs in the farmland pedosphere, these soil microorganisms may be exposed to long-term toxicity of various MP types. However, studies on this topic still are rare and knowledge is limited.

5.6 | Nematodes

Two studies on the effect of MPs on the nematode species *C. elegans* similarly reported decreased reproduction rates and altered transcriptional pathways identified through RNA interference screenings (Kim et al., 2020; Schöpfer et al., 2020). Reduction of nematode abundance and change in the community composition of soil fauna at high MP concentrations was reported in a study by Lin et al. (2020). A laboratory study by Mueller et al. (2020) showed various impacts on bacterial-feeding nematode species (*C. elegans*, *Acrobeloides nanus*, and *Plectus acuminatus*) by exposing them to PS beads. However, the nematodes response to MPs was species depended. Among the species *A. nanus* populations grew faster in presence of PS, whereas other species showed slower reproduction. Therefore, MPs have potential to alter nematodes population dynamics, resulting in an impact on the food web (Mueller et al. 2020; Schöpfer et al., 2020).

5.7 | Earthworms

Earthworms are perhaps the best studied organisms for MPs' effect on soil biology. Generally, there are three common points among most studies: (1) earthworms actively transport MPs via ingestion and subsequent defecation, with possible further fragmentation and gut tissue damages; (2) ingestion is size dependent; and (3) effects on earthworm fitness are not consistent but highly dependent on MPs characteristics and environment (Cao et al., 2017; Lahive et al., 2019; Rodriguez-Seijo et al., 2017). Studies by Huerta Lwanga et al. (2016, 2017) found various implications for the effect of MPs on the earthworm species *L. terrestris*. The worms were exposed to different concentrations of LDPE surface litter. Organisms' biomass, burrow formation and characteristics, biogenic transport, and particle size were measured. Overall, *L. terrestris* was observed to actively and selectively transport MPs from the surface to deeper soil layers, a result that was confirmed by Rillig et al. (2017). The small diameter of the worm's mouth (approximately 3 mm) prevents it from ingesting bigger particles (Rodriguez-Seijo et al., 2017). Smaller particles were found deeper in the soil, and fractions ≤50 μm increased by 65% in the burrows compared with the soil surface and in the casts. Earthworm (*L. terrestris*) mortality increased by 8–25% at ≥28% MPs concentration in 60 days of exposure, and the growth rate was significantly reduced at 28, 45, and 60% of MPs concentration in the litter at a mesocosm experiments (Huerta Lwanga et al., 2017). Similar findings are reported by Cao et al. (2017) for *E. fetida*, whose growth was inhibited and mortality increased at PS–MPs concentrations of 1 and 2% with a size of 58 μm. Earthworm *E. crypticus* was also reported to ingest nylon particles, with a significant effect on reproduction in a dose-dependent manner where smaller sizes and higher concentrations MPs had a more substantial impact (Lahive et al., 2019).

Toxicological studies have reported gut disturbances and altered oxidative stress systems in various earthworm species. The laboratory study by D. Zhu, Chen et al. (2018) on *E. crypticus* reported weight reduction for concentrations of NPs at 10% of soil dry weight and a significant shift in the gut microbiome, with a decrease in relative abundances of microbe families, which contribute to nitrogen cycling. Histopathological analysis of *E. andrei* exposed to increasing concentrations of PE MPs reported atrophy or detachment of the gut epithelium. Other significant histopathological alterations such as congestion, fibrosis, and inflammatory infiltrates were also reported (Revel et al., 2018; Schöpfer et al., 2020). In a similar study, oxidative stress, energy metabolism, and molecular responses of *E. fetida* were measured after exposure to different concentrations of PE for 28 days. Significant effects were found in an unbalanced oxidative stress system, indicating that MPs levels ≥500 mg MPs kg^{-1} soil dry weight may constitute a threshold (Rodriguez-Seijo et al., 2017). A laboratory study by Wang, Coffin et al. (2019) measured the toxicological effects of MPs on *E. fetida* for 14 days at increasing concentrations. Contrary to the previous study, no discernible effect on oxidative stress levels was detected, and MPs did not significantly contribute to contaminant bioaccumulation.

E. fetida, exposed to PS particles for 14 days, displayed significant bioaccumulation of PS. The histopathological analysis indicated gut damage and oxidative stress after exposure, and the comet assay suggested the damage extended to the DNA as well (Jiang et al., 2019). The results by Prendergast-Miller et al. (2019), who studied the effect of PS microfibers on *L. terrestris*, also supported toxicological impacts on earthworms. Though no increased mortality was found, there was evidence of modified casting behavior, possibly caused by variations in stress biomarkers. The variety of results highlights the complexity of these measurements, as metabolic pathways are many, and stress responses may include different molecular mechanisms. It was also stated by Sanchez-Hernandez et al. (2020) that to better understand the effect of MPs on earthworms (and vice versa), future research needs to include more earthworm species, such as the endogeic species *Aporrectodea* spp.

Moreover, NPs may have an even more significant effect on the organism than MPs. Studies by Van der Ploeg et al. (2011, 2013) found that fullerene C60 nanoparticles affected cocoon production, juvenile growth rate, and mortality of *L. rubellus*, with sub-lethal effects on tissue pathologies and altered gene expression. Addressing a global issue, the improper disposal of face masks in the light of the COVID-19 pandemic may also affect the soil ecosystem. A recent laboratory study by Kwak and An (2021) investigated the effects of meltblown face mask filters (MB filters) on soil species (earthworm and springtail). The results of MB filter at a high concentration of 1000 mg kg^{-1} dry soil showed inhabitation of spermatogenesis in male reproductive tissues of earthworms, growth decrease in collembolan, and reduction of intracellular esterase activity in earthworm coelomocytes (Kwak and An, 2021).

Overall, MPs' impacts on earthworms vary from increased mortality to reduced growth and reproduction, damaged gut linen, and possibly damaged DNA. Importantly, worms transport MPs particles into deeper soil horizons, distributing them over the entire soil profile and thereby making them bioavailable to other organisms. Earthworms are fundamental for agricultural soils,' damage to this group can impact crop growth (Brown et al., 2004; Forey et al., 2011).

5.8 | Chemical interactions of MPs and soil microbial activity

Much of the residue derived from plastics used in the agricultural system (e.g., plastic mulches, greenhouse covers, silage baling, containers, plastic shelters, packaging, and netting) can indirectly affect microbial communities. Mulches can alter soil microclimate and atmospheric boundary conditions by acting as a barrier, trapping moisture, reducing evaporation and gas exchange, and increasing temperature by changing the albedo. Modifying key abiotic factors can impact microbial activity, relative abundance, community structure, and grazing activity (Kasirajan & Ngouajio, 2012; Schirmel et al., 2018). Plastic mulches can significantly decrease taxonomic richness, with roughly a 50% decrease in the abundance within the analyzed taxa (Schirmel et al., 2018). Mulches are fundamental in nitrogen and carbon cycling by modulating SOM decomposition (Heijboer et al., 2018). SOM is critical to soil integrity and functioning, affecting crop growth, while crop yield will depend on anthropogenic additives, fertilizers, pesticides, and herbicides and eventually substantially impact food security. Decreased soil microbial activity was positively correlated to mulch residues and phthalates (PAEs) concentration, substances such as plasticizers, which can alter gene expression in mammals (Singh & Li, 2012) and disturb neonatal hormones (Sharpe, 2001). Furthermore, the potential of adverse impacts of MPs/NPs can be generated by plastics intrinsic toxicity (e.g., physical damage), chemical composition (e.g., leaching of additives such as PCBs: polychlorinated biphenyls; DDE: p,p′-dichlorodiphenyl dichloroethene; PAHs: polyaromatic hydrocarbons; PBDEs: polybrominated diphenyl ethers; NP: nonylphenol; OP: octylphenol; BP: bisphenol), and MPs ability to adsorb and release environmental pollutants into the soil organisms (Bouwmeester et al., 2015; Revel et al., 2018).

5.9 | Microbial enzyme activity and biodegradation of plastic debris

In soil, exposed plastic debris becomes brittle due to UV radiation, mechanical abrasion, weathering, and interaction with fauna and fragmentation into smaller particles, called secondary MPs (Brandes et al., 2020; Yu & Flury, 2021). UV radiation causes photodegradation by UV-A (≈315–400 nm) and UV-B (≈295–315 nm) radiation, both responsible for photolysis and photo-oxidation. Thermal oxidation is facilitated by absorbing visible light (380–750 nm) and infrared light (760–2500 nm). The MaPs debris found in agricultural fields will break down into MPs and NPs remaining trapped in soil for many years (Liu et al., 2014). Degradation by abiotic factors decreases the molecular and gravimetric weight of the polymer, allowing microorganisms to penetrate more quickly through the cell membrane and functioning extra- and intracellular enzymes (Sivan, 2011). Biodegradation is defined as the process by which organic substances are decomposed by microorganisms (mainly aerobic bacteria) into substances such as carbon dioxide, water and ammonia. However, even though there are some organisms known to decrease the gravimetric and molecular weight of the polymers, the process of total degradation of a plastic fragment under environmental conditions can take hundreds of years, and it is still not clear if it can completely degrade (Jin et al., 2022; Palmisano & Pettigrew, 1992).

Biodegradation of plastic polymers is a multistep process. The prerequisites include plastic deterioration, microbial colonization, production of polymer-degrading exoenzymes, and mineralization (Sanchez-Hernandez et al., 2020). Given their bioturbation and organic waste decomposition (vermicomposting), earthworms can support microbial colonization, deterioration, mineralization, and degradation by exoenzymes (Sanchez-Hernandez et al., 2020). They would therefore help other organisms to decompose polymers and thus accelerate decay. Both bacteria and fungi have been shown to degrade PE, one of the most used plastic types in agriculture. Low-weight products derived from oxidation are metabolized by bacteria, which form a biofilm around the polymer (Bonhomme et al., 2003; OECD Directorate, 2002). Studies have taken different approaches to determine the biodegradability of polymers. Many have used pure cultures under laboratory conditions to treat the polymer. Genera from the most common soil strains such as *Pseudomonas*, *Brevibacillus*, *Bacillus*, *Rhodococcus*, *Staphylococcus*, and *Streptomyces* have been reported to decrease the molecular weight of HDPE and LDPE (Bhardwaj et al., 2013; Hadad et al., 2005). Three species of *Streptomyces*, a common strain found in soils, have been shown to produce extracellular enzymes that change PE's mechanical properties after a 3-week incubation (Lee et al., 1991; Pometto et al., 1992). *Pseudomonas chlororaphis* was found to degrade polyester–polyurethane with extracellular enzymes, using plastic as the sole carbon source (Howard et al., 1999). Experiments on bacterial and fungal degradation of polymers concerning SOM content were conducted by Nowak et al. (2011). Results showed that (1) soils with higher SOM content more rapidly degraded plastic and (2) both the bacteria and the fungi efficiently colonized PE. Studies investigating a limited number of pure strains have determined vital metabolic pathways. First, bacteria form a biofilm around the polymer, and subsequently, extracellular enzymes start the degradation process. These enzymes are part of the depolymerase family, which break down complex chains into short-chain molecules and monomers that then can be used by microorganisms as a carbon source or will be further broken down and mineralized by other organisms. Accordingly, studies on microorganism communities' capacity to degrade polymers have shown that (1) molecular weight, crystallinity, and physical forms determine the amount of degradation of the polymer; (2) environmental conditions play a significant role in determining the community found in the soil; (3) aerobic and anaerobic consortia are involved in the deterioration; and (4) pretreating the polymer, by UV or heat exposure, significantly increases its biodegradability (Gu, 2003; Yuan et al., 2020).

Fungi also play an essential role in the degradation chain of polymers. *Phanerochaete chrysosporium* (Orhan & Büyükgüngör, 2000), different strains from the *Aspergillus* genus (Das & Kumar, 2015) and *Fusarium* genus have been reported to decrease gravimetrical and molecular weights of polymers. Bacterial degradation can decrease the polymer's weight by less than 1% after approximately 75 days and 225 days, according to the study conducted by Nowak et al. (2011). After 1 year under laboratory conditions, thermally heated HDPE and LDPE lost about twice the weight when treated with *Bacillus sphericus*, compared with untreated samples (Sudhakar et al., 2008). Nitrogen and phosphorous fertilizers are applied in high doses in some agricultural soils (S. Zhang et al., 2020). They have been shown to increase LDPE degradation and increase biodiversity and abundance of several predominant bacterial and fungi taxa, possibly facilitating the increased degradation of the polymer (B. Zhang et al., 2020). Stable conditions do not occur under natural conditions in the field, which drastically decreases the actual weight loss percentage and increases the polymer's turnover time. Hence, it is necessary to further investigate the factors involved in the agricultural management and assess long-term MPs contamination consequences.

6 | MPs TOXICITY EFFECTS ON CROP GROWTH IN FARMLANDS

Plastics in agricultural soils is of current concern due to their potential effects on the soil–plant system (Piehl et al., 2018). With a meta-analysis of different studies in China, Gao, Yan et al. (2019) showed a yield decrease with an increasing amount of plastic residue; when the plastic debris was >240 kg ha^{-1} (≈0.15 g kg^{-1}) in fields using plastic mulch. One explanation is the obstruction of root growth due to MaPs debris in soil (Y. Qi et al., 2018). However, the effects of MaPs, MPs, and NPs cannot be easily differentiated. Laboratory studies showed that the expected impacts of MPs debris on plant growth would differ based on the size, shape, and type of polymer of the MPs (Ebere et al., 2019).

A field study by de Souza Machado et al. (2019) reported that *Allium fistulosum* (spring onion) responded to different types of MPs, demonstrating that the root biomass was significantly increased in the presence of polyester fibers and PS but not by HDPE particles. However, in a similar study, plant height, shoot biomass, and leaf area of wheat (*Triticum aestivum* L.) and juvenile lime tree (*Citrus aurantium* L.) were reduced in MP treatments (Boots et al., 2019; Li et al., 1999; Verla et al., 2020).

Furthermore, Bosker et al. (2019) reported that the germination rate of cress (dicotylodon *Lepidium sativum* L.) was significantly reduced after 8 h of exposure for all sizes of NPs (50, 500, and 4800 nm) with the increasing adverse impact of particle sizes. However, the impact of NPs and MPs on seed germination disappeared after 24 h of exposure, and germination reached close to 100% regardless of the plastic size or exposure concentration (Bosker et al., 2019).

As previously discussed, plastic debris have been reported to alter soil physical, chemical, and biological properties, leading to indirect effects on plant growth (Figure 2). However, it should be noted that the impact of MPs on soil physicochemical properties and biota is still not fully understood and predictable. Hence, the consequences for plants are also not clarified. Moreover, soil biota plays a major role in soil fertility (Usman et al., 2016). Y. Qi et al. (2019) showed that biodegradable plastic debris had a more substantial impact on the composition of wheat rhizosphere bacterial communities and was associated with lower plant biomass than LDPE debris. The study suggests that biodegradable residues could reduce plant biomass by releasing toxic compounds in the soil solution and modifying the soil microbiome (Y. Qi et al., 2019).

More specifically, NPs can enter the plant body from the soil and accumulate in tissues and cells (Jiang et al., 2019; Li et al., 2019; Urbina et al., 2020). For instance, Bandmann et al. (2012) observed NPs in tobacco BY-2 cells. The NPs involved clathrin-dependent and clathrin-independent endocytosis. Jiang et al. (2019) also demonstrated that under oxidative damage, PS-MPs could accumulate in *V. faba* root. Similarly, Jiang et al. (2019) observed a decrease in biomass and catalase enzyme activity of *V. faba* roots under 5 μm PS–MPs, while superoxide dismutase and peroxidase enzymes activity significantly increased. Under 100 nm PS–MPs exposure, a significant growth reduction was observed only at the highest concentration (100 mg L^{-1}). Micronucleus tests and antioxidative enzyme activities showed that 100 nm PS-MPs induce greater genotoxic and oxidative damage to (*V. faba*) than 5 μm PS–MPs (Figure 3). Additionally, Sun et al. (2020) demonstrated that positively and negatively charged NPs could accumulate in *Arabidopsis thaliana* with different effects depending on the NPs surface charge: NPs with positive surface charge tended to form aggregates, promoted by the growth medium and root exudates, and therefore were less absorbed than negatively charged NPs. Nevertheless, positively charged NPs induced a higher accumulation of reactive oxygen species and inhibited plant growth and seedling development more strongly than negatively charged NPs. By contrast, the negatively charged NPs were observed frequently in the apoplast and xylem. Results of a study by Sun et al. (2022) revealed the detrimental effects of NPs (PSNPs-NH2) on the molecular photosynthesis system in corn (*Zea mays* L.)

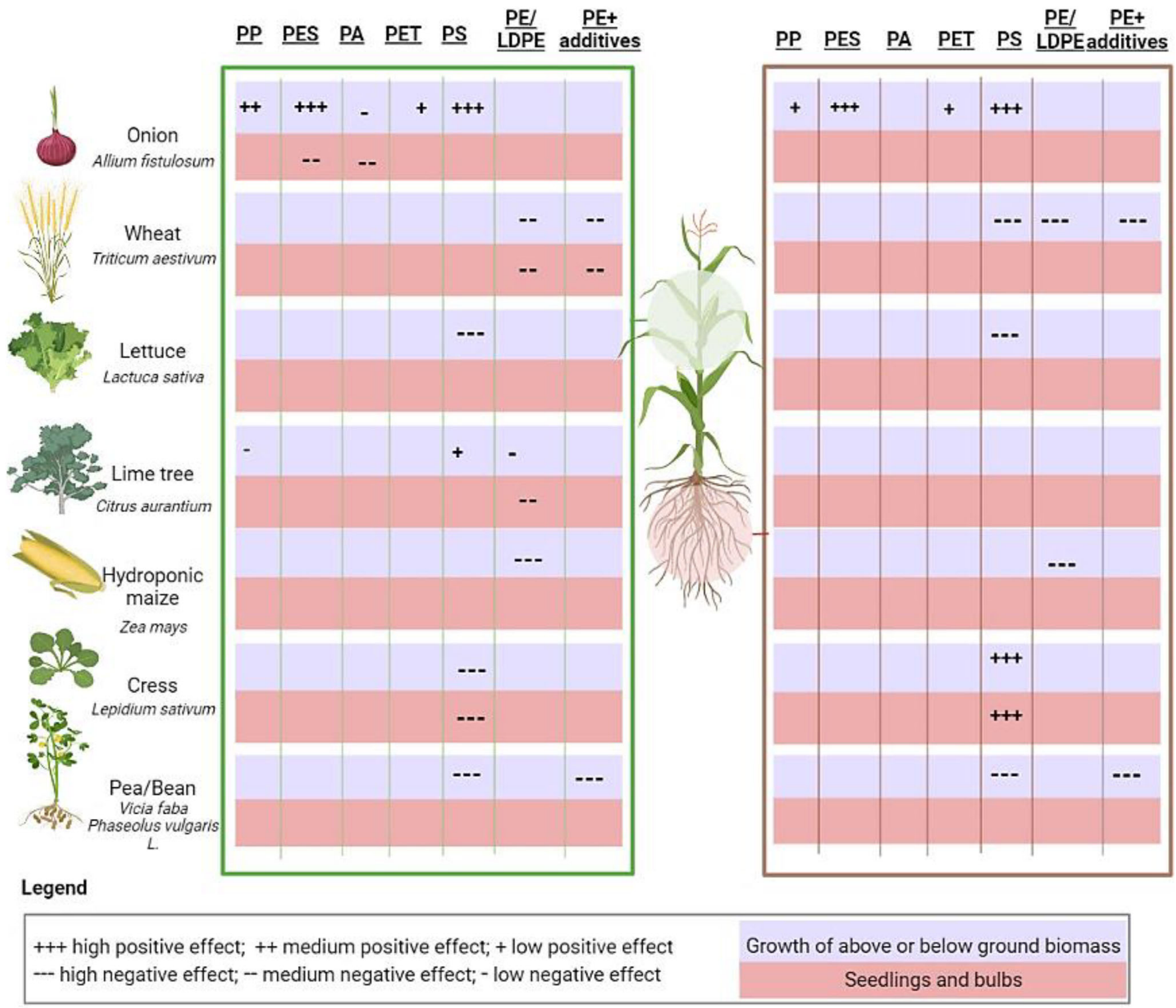

FIGURE 2 Examples of microplastic effects on various parts of plants. A description of applied concentrations of polymers per plant is available in the Supporting Information.

through the impairment of the photosystem II efficiency via the down-regulation of the transporter D1 protein, resulting in a more significant inhibitory impact on plant growth. The results of these studies highlight the human exposure pathways to MPs through the consumption of contaminated crops and emphasize the need for new management strategies to control the release of MPs, which is indicated by the MPs potential influences on food safety, crop plants, and human health (Silva et al., 2021; Wang et al., 2021).

Despite growth parameter alterations with MPs, the possible pathways of MPs affecting plants' function are still unclear. One possible pathway would be the immobilization of nutrients by organic compounds originating from degradation (Martin-Closas et al., 2014; B. Zhang et al., 2020).

A field study by Li et al. (2019) displayed the uptake and distribution of PS microbeads (0.2 and 1.0 μm) via lettuce roots. The authors used fluorescent markers to track PS microbeads in lettuce tissues, which have been trapped in the root cap mucilage as it has a higher number of hydrated polysaccharides. Confocal images indicated that microbeads mainly passed through the intercellular area and via the apoplastic and vascular systems (PS size 0.2 μm). Microbeads relocated from roots to stem and leaves with the transpiration stream (Li et al., 2019). Sun et al. (2021) observed a different pathway of NPs uptake (leaf to root translocation) in maize. The findings of that study indicate that PS–NPs entered the plant vascular system via the stomatal pathway, moving down to the roots. The positive surface charge (PS–NH_2) allowed aggregation on the leaf surface, hindering the PS movement to the roots. The positive charge of PS showed higher inhibitory effects on leaf photosynthesis and more vigorous agitation for antioxidant systems activation. The PS with a negative charge (PS–COOH) are transferred faster to the root area (Sun et al., 2021). NPs have tremendous effects on biochemical enzymes, the antioxidant system, electrolyte leakage, it blocks cell wall pores and causes oxidative

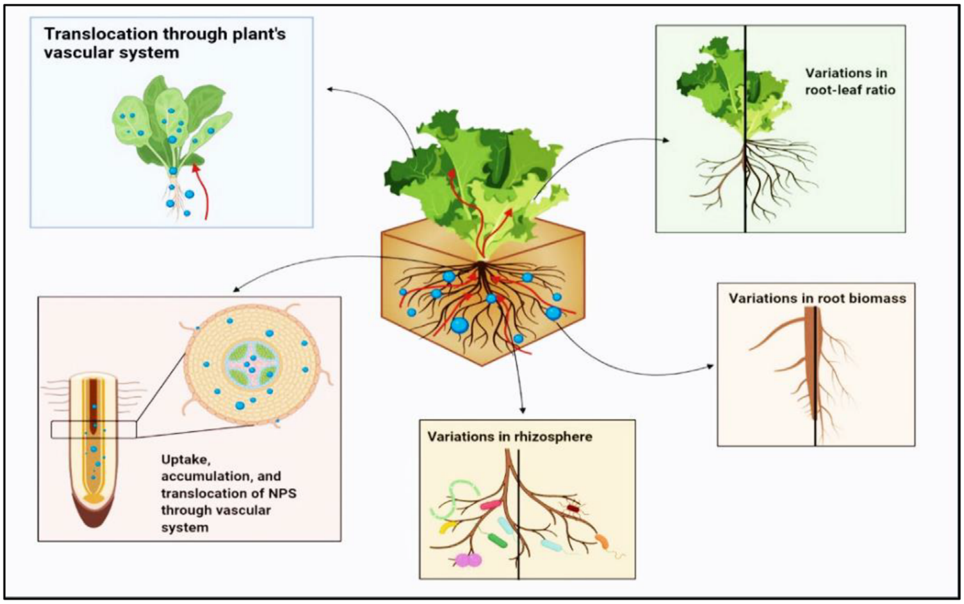

FIGURE 3 Schematic figure of possible uptake mechanisms and effects on plant growth of MPs and NPs by plants

damages of plants (Adeel et al., 2021; Azeem et al., 2021; Gao, Liu et al., 2019; Sun et al., 2021).

7 | ACCUMULATION OF MPs IN THE FOOD CHAIN

MPs ingested by organisms can accumulate in the food chain (Figure 4). The study of exposure of terrestrial animals to MPs showed that many organisms do ingest plastics. For example, MPs concentrations increased from soil (0.87 $\pm$ 1.9 particles g^{-1}) to earthworm casts (14.8 $\pm$ 28.8 particles g^{-1}) to chicken feces (129.8 $\pm$ 82.3 particles g^{-1}) in home gardens in Southeast Mexico (Huerta Lwanga et al. 2017). Zhao et al. (2016) identified similar plastic content in 17 wild birds around Shanghai with an average of 22.8 $\pm$ 33.4 particles per bird stomach (including proventriculus and gizzard) and their esophagus and intestines. In addition, MPs have been found in the digestive systems of larger animals (Mekuanint et al., 2017). For example, Beriot et al. (2021) identified an average of 1000 particles kg^{-1} of feces of sheep grazing in vegetable fields where plastic mulch has been used in the south of Spain. Calcium and phosphorous deficiency and insufficient nutritional supplementation are identified as predisposing factors for plastic ingestion by roaming livestock (Priyanka & Dey, 2018). Potential effects of plastic ingestion on organisms include blockage of the intestinal tract, inhibition of gastric enzyme secretion, reduced feeding stimuli, decreased steroid hormone levels, delays in ovulation, and even failure to reproduce (Enyoh et al., 2020). Additionally, NPs were observed to be transported from the soil to plants and snails, affecting their growth, locomotor activity, and intestinal microbiota viability (Chae & An, 2018).

However, ecotoxicological effects at environmental concentrations are still not well studied (Ng et al., 2018; Unrine et al., 2012). The contamination occurring in agricultural soils is likely to lead to human exposure to MPs and NPs through wind transport or contamination of the food and the freshwater. Ingestion is considered the primary route of human exposure to MPs (Galloway, 2015). The estimated MP intake is 39,000–52,000 particles person^{-1} year^{-1} (Cox et al., 2019).

8 | TRANSPORT AND RETENTION OF MPs IN SOIL AND TO THE GROUNDWATER

MPs have been reported to affect water percolation in soil. However, research is critically needed to determine how deep MPs will migrate in soil over time. A recent field study from central Germany suggested that MPs could be transported into soil layers as deep as 1 m (Weber & Opp, 2020). Leaching is a critical process of transporting MPs to groundwater (O'Connor et al., 2019). The transport of plastic debris through biopores has been identified as a possible mechanism for groundwater contamination; however, knowledge regarding this phenomenon is scarce. A study by He et al. (2018) provides first evidence on the vertical migration of plastic in agricultural soils toward aquifer systems, especially for MPs, analogously to the well-known migration of natural particles in the colloids (e.g., pesticides). Hence, the leaching process presumably poses a significant risk for underlying aquifers, floodplains, and drinking water supplies (e.g., groundwater resources below farmlands) to be contaminated by plastic and pesticides (Wanner, 2021). Various studies have shown that the mobility of plastics in agricultural soils can be variable, depending on different conditions (e.g., soil water saturation, aggregation, and deposition of MPs, the number of wet–dry cycles, earthworms, microarthropods, and agricultural activities) (Maaß et al., 2017; Huerta-Lwanga et al., 2017; Rodriguez-Seijo et al., 2018; Yan et al., 2020). The various tillage practices affect different soil layers and the depth to which MP can be

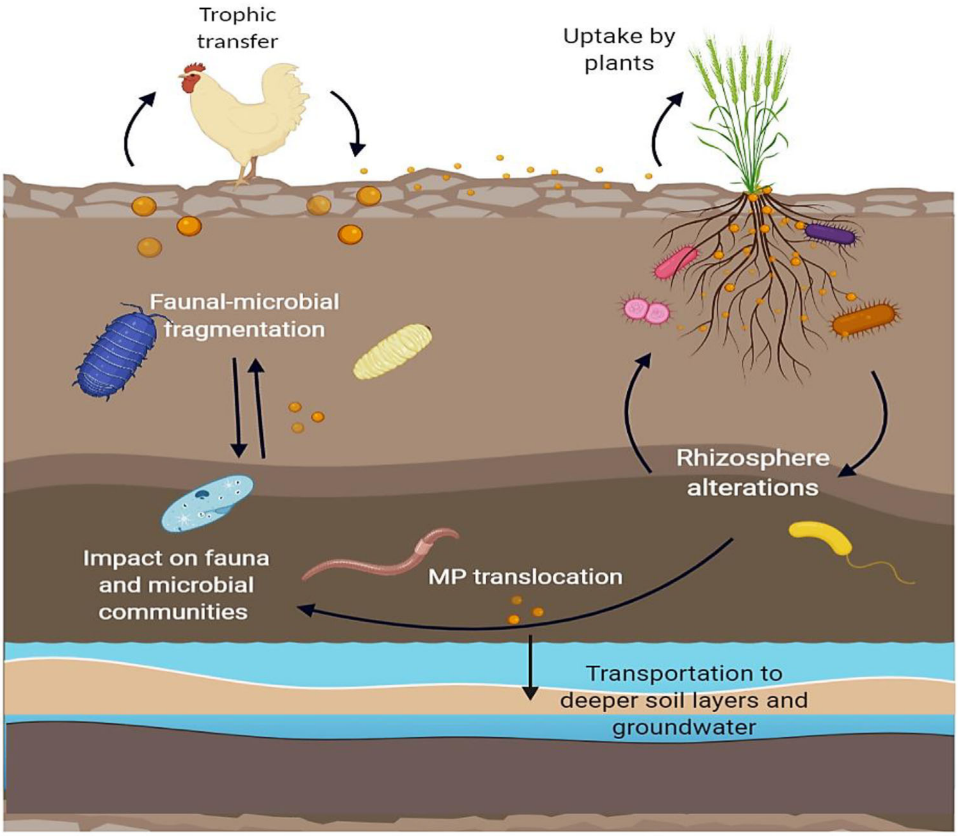

FIGURE 4 Trophic transfer of MPs in soil and groundwater

incorporated (Rillig et al., 2017). Rehm and Fiener (2020) showed that MPs have a high transport potential in the soil, with dependence on retention time and the soil texture.

The transport and retention of MPs in soil are mainly related to the suspension physicochemical characteristics (e.g., particle and grain sizes, ionic strength and cation type, flow rate, and other coexisted colloids) (Wu, Lyu et al., 2020). Most studies about the transport of MPs into the soil column were done with specific substances (e.g., well-defined quartz, sand, or glass beads), which can only represent the complex natural soil conditions to a very limited extent.

There are three physical transport mechanisms crucial for microscale contact: Brownian diffusion, direct mechanisms, and gravitational sedimentation (Xu et al., 2016). The main underlying mechanisms (e.g., bioturbation, attachment, and detachment of the bulk soil, homo aggregation; with similar material and hetero aggregation; for particles of opposite surface charge, straining and size exclusion, interactions with dissolved organic matter, and steric stabilization, and interactions in unsaturated porous media) have the potential to influence particle mobility (Cornelis et al., 2014; Fujita & Kobayashi, 2016; Mitrano et al., 2019; Shen et al., 2012). With precipitation and irrigation in farmlands, plastic residues may be transported into greater depths with the percolating water through soil macropores and finally could end up in the groundwater.

The scarcity of experimental data to confirm these findings emphasizes the need of a theoretical assessment of MPs transport based on hydrological/sediment transport catchment models. These models could explain the potential for soils to effectively retain and store MPs and NPs (Hurley et al., 2018). Besides, the investigation of vertical transport of MPs, risk assessments for plant uptake, and significantly more data on spatial dependencies of plastic contamination can further improve the knowledge on MPs contaminated soils (Weber & Opp, 2020).

9 | CONCLUSION AND FUTURE RESEARCH PERSPECTIVES

This review paper summarized the existing research on MPs and NPs in agricultural soils, outlined the potential MPs sources, and discussed their impacts on soil properties, microbiome, and plant growth parameters. Various sources of primary and secondary plastic residues enter the soil environment in farmlands. Most of the available studies on MPs' impacts on agricultural soils mainly focused on laboratory or short-term experiments with MPs concentrations between 0.001 and 2% (w/w) per dry soil weight in farmlands. However, the long-term effects of accumulated MPs in soils and groundwater systems at various concentrations are still unclear. Besides, water flow and bioturbation facilitate MPs and NPs translocation in the soil. In general, the fate and transport of MPs are connected to environmental factors (e.g., UV irradiation, hydrolysis reactions, and temperature).

Additionally, the surface of the plastic becomes more hydrophilic by hydrolytic reactions. However, available data on these phenomena are still rare. Furthermore, MPs may enter the food chain via plant root uptake, microorganisms, and animals. In farmland ecosystems, the effects of MPs on soil properties, soil environment, and plant production have potential consequences for human health by accumulating MPs and NPs and harmful compounds in the tissue of plants. This review identified a number of open research questions (Table S1), which need multidisciplinary research on the field scale and modelling of MPs pollution potential in agroecosystems. Generally, research on MPs contamination of agricultural soils is a comparatively new field. Results so far indicate MPs contamination to be a severe hazard for soil and human health. This emphasizes the need for further and fast research, especially on possible entry sources. Further, research should be extended to soils in general as non-agricultural soil may also be affected (e.g., by wind transport) with so far unknown consequences. The major topics that should be included in future research are:

1. Design and develop experiments to survey the potential toxicity of particles connected with changes in MPs surface structures (e.g., physical abrasions, UV irradiation, chemical interactions, sorption potential, and biological attack).
2. Design in situ experiments or using undisturbed soil columns to better understand MPs' transport and movement rates of pristine and aged polymers through the vadose zone toward the aquifer.
3. Investigate MPs features (e.g., MPs chemical, structural and physical characteristics), and their interaction with soil aggregates, assessment of MPs and NPs fate in soil porous media and water bodies, and their possible entering pathway to the food chain and how human health is affected.
4. Develop a fast and accurate detection method for MPs <100 μm in soil and underground water at reasonable costs.

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DATA AVAILABILITY STATEMENT

Data sharing not applicable to this article as no datasets were generated or analyzed during the current study.

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SUPPORTING INFORMATION

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