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Review

Ecotoxicological effects of micro- and nanoplastics on terrestrial food web from plants to human beings

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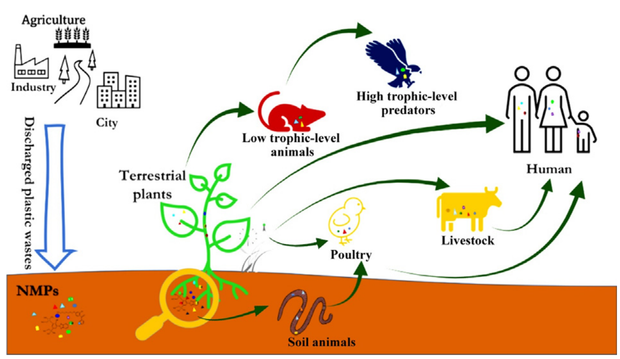
HIGHLIGHTS

- Uptake of MNPs occurs in both terrestrial flora and fauna.
- Exposure to MNPs induces varying effects on terrestrial biota.
- The effects of MNPs on bioavailability of co-present chemicals are debatable.
- There is evidence of trophic transfer of MNPs in terrestrial food web.
- Data regarding the intake and impacts of MNPs in humans are scare.

GRAPHICAL ABSTRACT



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ABSTRACT

Micro- and nanoplastics (MNPs) are present in almost all environmental compartments. Terrestrial soils are major environmental reservoirs for MNPs, but the ecotoxicological effects of MNPs on terrestrial biota remain relatively understudied. In this review, we collated findings of previous research on the uptake and impact of MNPs in terrestrial organisms, including flora, fauna, and human beings. Terrestrial plants can take up MNPs via the roots or leaves and translocate them to other parts. MNPs have been detected in the gastrointestinal tracts or feces of many terrestrial animals, including some high trophic-level predators, indicating the incidence of direct ingestion or trophic transfer of MNPs. The presence of MNPs in food items and human feces combines to verify human intake of MNPs via the dietary pathway. Exposure to MNPs can cause diverse effects on terrestrial organisms, including alterations in growth performance, oxidative stress, metabolic disturbance, cytotoxicity, genotoxicity, and mortality. The biological internalization and impact of MNPs are influenced by the physicochemical properties of MNPs (e.g., particle size, polymer type, surface chemistry, and exposure concentrations) and the physiology of the species. MNPs can also affect the bioavailability of co-occurring intrinsic or extrinsic contaminants to terrestrial biota, but their specific role is under dispute. Finally, we underlined the current research gaps and proposed several priorities for future studies.

Contents

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1. Introduction

The extensive use of plastics in modern daily life has boosted the manufacture of these polymeric materials in the past decades (PlasticsEurope, 2021), leading to the increased release of plastic waste into the environment. Terrestrial soils are the main recipients of plastic wastes, with the annual load of plastic litter in soils being 4–23 times that in the oceans (Horton et al., 2017). Driven by the combined actions of biotic and abiotic processes, discarded plastic debris can undergo progressive fragmentation into smaller particulates, which constitute the major source of microplastics (MPs, 0.1 μm–5 mm) and nanoplastics (NPs, <0.1 μm) in the environment (da Costa et al., 2016; Jiang, 2018; Kwak and An, 2021). Micro- and nanoplastics (MNPs) can also be purposefully manufactured for specific use in some sectors, such as personal care and cosmetic products, electronics, and medicine, and enter the environment via direct release (Cheung and Fok, 2017; da Costa et al., 2016; Pohlmann et al., 2013). The presence of MPs has been documented in a variety of terrestrial systems, including farmlands, grasslands, woodlands, greenhouse fields, and flood plains (Ding et al., 2021; Scheurer and Bigalke, 2018; Zhang and Liu, 2018). In highly contaminated sites, soil concentrations of MPs can reach g/kg levels (Fuller and Gautam, 2016). Although the environmental occurrence of NPs remains largely unknown owing to the limitations of detection techniques, there is a universal consensus that these nano-sized particles may accumulate in the environment at considerably high concentrations (Amorim and Scott-Fordsmand, 2021; da Costa et al., 2016).

There are many pathways for MNPs to enter terrestrial soils, such as land application of sewage sludge and compost, irrigation with contaminated water, use of plastic mulching, and atmospheric deposition (Wang et al., 2020c; Weithmann et al., 2018; Yang et al., 2021). For instance, land application of sludge leads to an annual release of 63,000–430,000 tons of MPs into European farmlands (Nizzetto et al., 2016). The annual deposited mass of road-derived MPs on land is estimated to be 343 kt (Evangeliou et al., 2020). Because of their multiple sources, MNPs can markedly vary in size, shape, color, and chemical composition, leading to different impacts on soil properties and biota (de Souza Machado et al., 2019; Wang et al., 2020c). Furthermore, MNPs can serve as vectors for diverse intrinsic (e.g., plastic additives and monomers) and extrinsic (e.g., adhered chemicals and pathogens) contaminants, consequently affecting their dissemination in the environment and availability to biota (Koelmans et al., 2016; Mammo et al., 2020). The combined effects of MNPs and associated contaminants on terrestrial biota remain incompletely characterized.

A growing body of evidence suggests that terrestrial plants can take up MNPs and translocate them to the edible portions, providing a feasible route for MNPs to enter the terrestrial food web (Li et al., 2020a; Li et al., 2020b; Wang et al., 2021d). Ingestion of MNPs also occurs in many terrestrial animals via intentional or accidental swallowing (Cheng et al., 2021; Sherlock et al., 2021). Once inside the animal body, very tiny MNPs (e.g., <130 μm) can cross the epithelial barriers of the gastrointestinal tract into the circulatory system, leading to the dispersion of particles throughout the body (Haave et al., 2021; Wright and Kelly, 2017). Exposure to MNPs can cause deleterious effects on terrestrial flora and fauna, which may in turn impact the terrestrial food web (Baho et al., 2021; Deng et al., 2021; Sun et al., 2020; Wang et al., 2021d). Although humans are also susceptible to MNP exposure via pathways like dietary intake and inhalation (Du et al., 2020; Schwabl et al., 2019; Zhang et al., 2021a), information on the health impacts of MNPs on humans is scarce, which underscores the need to summarize current research progress and highlight priorities for future studies.

Given that soils are the major environmental reservoir for MNPs and have an important role in sustaining biodiversity, the risks of MNPs to terrestrial biota must be evaluated. Despite there have been several review papers summarizing the effects of MNPs on terrestrial biota, most of them focused on a specific group, such as flora, soil fauna, or humans (Cox et al., 2019; Rillig et al., 2019; Shi et al., 2021a; Vethaak and Legler, 2021; Wang et al., 2021c; Zhang et al., 2022). In this paper, we summarize the reported data regarding the uptake, translocation, trophic transfer, and biological impacts of MNPs in terrestrial biota at different trophic levels. By doing so, we provide a systematic evaluation of the ecotoxicological effects of MNPs on the terrestrial food web. Furthermore, based on an analysis of published research, current knowledge gaps are identified, and future research priorities are proposed.

2. Uptake and impacts of MNPs in terrestrial flora

2.1. Plant uptake of MNPs

As primary producers that present the major energy source for animals and humans, plants play a foundational role in the terrestrial food web. The prevalence of MNPs in terrestrial environments leads to inevitable interactions with plants. Until recently, there has been little evidence regarding the accumulation of MNPs in plants growing in field settings. The first clues were from Oliveri Conti et al. (2020), who investigated the presence of MNPs in tissues of edible fruits [i.e., apple (*Malus domestica*) and European pear (*Pyrus communis*)] and vegetables [i.e., lettuce (*Lactuca sativa*), broccoli (*Brassica oleracea italica*), carrot (*Daucus carota*), and potato (*Solanum tuberosum*)] purchased from the local markets in Catania, Italy and reported that the mean concentrations of MPs (1–4 μm) in fruits and vegetables were in the ranges of 190,000–196,000 and 51,000–126,000 particles/g fresh weight (fw), respectively. However, in the laboratory, root uptake and acropetal transport of MNPs in terrestrial plants have been well documented for a variety of species, including lettuce, carrot, cucumber, onion, wheat, rice, maize, bean, and thale cress, some of which are edible crops (Table 1). For instance, in a hydroponic study of cucumber (*Cucumis sativus* L.), Li et al. (2020b) found that polystyrene (PS) MPs (0.1–0.7 μm) could enter the plant roots and subsequently disperse to different aboveground parts, including the stems, leaves, flowers, and fruits. Accumulation of PS and polymethylmethacrylate (PMMA) MPs (≤ 2 μm) has also been observed in the roots, stems, and leaves of lettuce (*Lactuca sativa* L.) and wheat (*Triticum aestivum* L.) grown in soil and nutrient solutions (Li et al., 2020a). For root uptake, MNPs must cross a series of physiological layers (e.g., the epidermis, cortex, and endodermis) to reach the

Table 1
Root and foliar uptake of MNPs by terrestrial plants.

Plant species	MNPs			Location of MNPs in plant	Reference
	Polymer type	Particle size (μm)	Concentration		
Root uptake					
Carrot (*Daucus carota* L.)	PS	≤1	10–20 mg/L	Root & leaf	(Dong et al., 2021)
Cucumber (*Cucumis sativus* L.)	PS	0.1–0.7	50 mg/L	Root, stem, leaf, calyx, & fruit	(Li et al., 2020b)
Lettuce (*Lactuca sativa* L.) & wheat (*Triticum aestivum* L.)	PS & PMMA	≤2	0.5–50 mg/L or 150–500 mg/kg	Root, stem, & leaf	(Li et al., 2020a)
Rice (*Oryza sativa* L.)	PS	≤1	40 mg/L	Root, stem, & leaf veins	(Liu et al., 2022)
Broad bean (*Vicia faba*)	PS	0.1	10–100 mg/L	Root tips	(Jiang et al., 2019)
Mung bean (*Vigna radiata*)	PS	0.03	10–100 mg/kg	Plant vasculature & leaf	(Chae and An, 2020)
Thale cress (*Arabidopsis thaliana*)	PS-NH_2 & PS-COOH	0.2	10–100 mg/L	Root	(Sun et al., 2020)
Foliar uptake					
Maize (*Zea mays* L.)	PS-NH_2 & PS-COOH	0.02	2 μL of 1 mg/L PS solution per 0.3 cm^2 leaf surface	Root, stem, & leaf	(Sun et al., 2021)
Lettuce (*Lactuca sativa* L.)	PS	0.1	0.1–1 mg/L	Root & leaf	(Lian et al., 2021)

PS, polystyrene; PMMA, polymethylmethacrylate.

root xylem. Although the apoplastic movement of MNPs can be hindered at the endodermis by the Casparian strip, smaller particles are able to enter the root stele through regions where the Casparian strip is not fully developed (Li et al., 2020a). Once they have entered the root vasculature, MNPs can spread to the aboveground parts of the plant following the transpiration flow (Wang et al., 2021d; Zhang et al., 2019). Recently, foliar uptake and leaf-to-root translocation of PS NPs in maize (*Zea mays* L.) and lettuce (*Lactuca sativa* L.) have been documented, demonstrating that airborne MNPs can enter plants directly through the leaf stomata (Lian et al., 2021; Sun et al., 2021). MNPs internalized in terrestrial plants can enter primary consumers via food consumption, leading to the trophic transfer of MNPs (Chae and An, 2020).

The uptake and translocation of MNPs in terrestrial plants are influenced by multiple factors, including the physicochemical properties of MNPs, plant physiology, and environmental conditions. The particle size of MNPs is a key factor that determines whether they can be taken up by plants. Smaller plastic particles are typically more effective in crossing the physiological barriers for root uptake. This has been well demonstrated in a study where MNPs with a size of ≤2 μm entered the root xylem via a crack-entry mode and dispersed to the aboveground portions of plants, while ≥5 μm particles were barely absorbed (Li et al., 2020a). The surface charges of MNPs also influence their uptake by plants. Compared to amino-modified PS NPs (PS–NH_2), the carboxyl-modified analog (PS–COOH) was found to be more readily taken up by plant roots or leaves (Sun et al., 2021; Sun et al., 2020). Higher particle aggregation trends and electrostatic attraction to the negatively charged cell wall were considered as the primary reasons for the lower plant uptake of PS–NH_2. Until recently, laboratory-synthesized pristine PS beads have been the most widely used MNPs in plant uptake studies, while the uptake behavior of MNPs with different polymer types, shapes, and weathering status remains largely overlooked. Additionally, root uptake of MNPs is considered to be a passive process driven primarily by the transpiration stream (Li et al., 2020a; Wang et al., 2021d; Zhang et al., 2019). Therefore, factors that influence plant transpiration may also have consequences for the uptake of MNPs. For instance, increasing temperature or reducing the relative humidity can enhance plant uptake and translocation of MNPs (Li et al., 2020a), whereas exposure of plant roots to aquaporin inhibitors exerts a negative effect (Zhou et al., 2021). The effects of other transpiration-related factors, such as plant species, growth stages, organs, light strength, carbon dioxide concentrations, and drought, on the uptake and accumulation of MNPs in terrestrial plants deserve further study.

2.2. Effects of MNPs on terrestrial plants

Exposure to MNPs can cause a variety of effects on the physiological and biochemical indicators of terrestrial plants (Table 2). Compared to the MP-free control, short-term (6 days) exposure of garden cress (*Lepidium sativum* L.) to polyethylene (PE), polypropylene (PP), and polyvinylchloride (PVC) led to a 42.1–78.9% and 22.2–55.6% decrease in shoot height and leaf number, respectively (Pignattelli et al., 2020). Significant concentration-dependent decreases in root length, relative root elongation, and biomass were observed in hydroponically grown broad bean (*Vicia faba*) following exposure to 10–100 mg/L of 5 μm PS MPs (Jiang et al., 2019). The inhibitory effects of MNPs on plant growth performance have been observed in other species, including lettuce (*Lactuca sativa* L.) (Gao et al., 2021b), onion (*Allium cepa* L.) (Giorgetti et al., 2020), rice (*Oryza sativa* L.) (Wu et al., 2020), wheat (*Triticum aestivum* L.) (Qi et al., 2018), maize (*Zea mays* L.) (Urbina et al., 2020), and thale cress (*Arabidopsis thaliana*) (Sun et al., 2020). Reduced water and nutrient uptake due to changes in soil physicochemical and microbial properties after MNP addition and blockage of root extracellular channels due to MNP accumulation are the possible reasons for the MNP-related suppression of plant growth (de Souza Machado et al., 2019; Urbina et al., 2020; Yu et al., 2020a). Another common symptom induced by MNP exposure is oxidative stress responses, which are typically indicated by the increased production of reactive oxygen species (ROS) and/or enhanced activities of antioxidant enzymes (Sun et al., 2020). For instance, root treatment with PS MPs significantly increased the accumulation of superoxide anion (O_2^-) and hydrogen peroxide (H_2O_2) in roots and leaves of lettuce (*Lactuca sativa* L.) (Gao et al., 2021b). In maize (*Zea mays* L.), the activities of superoxide dismutase (SOD), peroxidase (POD), and catalase (CAT) increased by 99.2–200.9%, 63.4–109.8%, and 33.4–191.9%, respectively, after leaf exposure to PS-NH_2 and PS-COOH NPs (Sun et al., 2021). In cases when the production of ROS exceeds the scavenging capacity of antioxidant enzymes, accumulation of ROS may impair photosynthesis and disrupt metabolic processes (Asada, 1999; Choudhury et al., 2013; Dong et al., 2020b; Giorgetti et al., 2020; Wu et al., 2020), which could be another possible reason for the MNP-related reduction in plant growth and development. However, it seems that the effects of MNPs on plant growth performance are not monotonically negative. For instance, Hernandez-Arenas et al. (2021) found that low concentrations (<2782 particles/kg soil) of MNPs promoted the growth of tomato (*Lycopersicon esculentum* Mill.). In soils amended with polyester (PET) fibers and PS fragments, the root growth of spring onion (*Allium fistulosum*) was significantly enhanced, while polyamide (PA) amendment fostered the growth of aerial parts (de Souza Machado et al., 2019). Improvements in some soil properties (e.g., aeration, moisture, and microbial activity) can be plausible explanations for the positive effects of MNP amendment on plant performance parameters (de Souza Machado et al., 2019). In addition, exposure to MNPs can induce cytotoxicity and genotoxicity in terrestrial plants, as evidenced by decreased mitotic index, increased micronucleus frequency, and occurrence of abnormal mitoses (Giorgetti et al., 2020; Jiang et al., 2019). Oxidative stress and impairment of antioxidant systems

Table 2
Effects of MNPs on terrestrial plants.

Plant species	MNPs			Co-present contaminants	Effects	Reference
	Polymer type	Particle size	Concentration			
MNPs alone						
Wheat (*Triticum aestivum* L.)	LDPE	<1 mm	1% (w/w in soil)	/	Decrease in plant growth and biomass.	(Qi et al., 2018)
Rice (*Oryza sativa* L.)	PS	20 nm	10–100 mg/L	/	Oxidative stress; Decrease in root length; Alternations in root metabolism processes.	(Zhou et al., 2021)
Maize (*Zea mays* L.)	PE	3 μm	0.0125–100 mg/L	/	Decrease in biomass and transpiration.	(Urbina et al., 2020)
Broad bean (*Vicia faba*)	PS	0.1–5 μm	10–100 mg/L	/	Decrease in root length and plant biomass; Decrease in cell proliferation; Oxidative stress; Genotoxicity.	(Jiang et al., 2019)
Garden cress (*Lepidium sativum* L.)	PE, PP, & PVC	≤130 μm	0.02% (w/w in soil)	/	Decrease in seed germination and plant biomass; Oxidative stress.	(Pignattelli et al., 2020)
Spring onion (*Allium fistulosum*)	PA, PS, HDPE, PP, & PET	<5 mm	0.2–2.0% (w/w in soil)	/	Polymer-dependent changes in plant total biomass, root traits, and leaf elemental composition.	(de Souza Machado et al., 2019)
Onion (*Allium cepa* L.)	PS	20–190 nm	0.01–1.0 g/L	/	Oxidative stress; Cytological abnormalities; Genotoxicity.	(Giorgetti et al., 2020)
Festuca brevipila, *Holcus lanatus*, *Calamagrostis epigejos*, *Achillea millefolium*, *Hieracium pilosella*, *Plantago lanceolata*, & *Potentilla argentea*	PET	1.3 mm	4 g/kg	/	Alterations in plant community structure; Decrease in community shoot to root ratio.	(Lozano and Rillig, 2020)
MNPs with other contaminants						
Rice (*O. sativa* L.)	PS & PTFE	10 μm	0.04–0.2 g/L	As (1.6–4 mg/L)	Decrease in plant biomass and photosynthesis; Reduced uptake of As; Oxidative stress.	(Dong et al., 2020b)
Lettuce (*Lactuca sativa* L.)	PE	10–500 μm	0.1–10% (w/w in soil)	Cd (0.5–4.4 mg/kg)	Decrease in plant biomass; Increase in plant uptake of Cd	(Wang et al., 2021a)
Lettuce (*L. sativa* L.)	PE	~23 μm	0.25–1 g/L	DBP (5 mg/L)	Decrease in root growth and activity; Damage in cell structure; Oxidative stress; Reduced uptake of DBP.	(Gao et al., 2021a)
Soybean (*Glycine* max L. *Merrill*)	PS	0.1–100 μm	10 mg/kg	Phe (1 mg/kg)	Oxidative damage; Decrease in root activity; Genotoxicity; Reduced uptake of Phe.	(Xu et al., 2021)
Wheat (*Triticum aestivum* L.)	PE	200–250 μm	0.5–8% (w/w in soil)	Phe (100 mg/kg)	Decrease in shoot height; Reduced uptake of Phe.	(Liu et al., 2021)

PE, polyethylene; LDPE, low-density polyethylene; HDPE, high-density polyethylene; PS, polystyrene; PP, polypropylene; PVC, polyvinyl chloride; PA, polyamide; PET, polyester; PTFE, polytetrafluoroethylene; As, arsenic; Cd, cadmium; DBP, dibutyl phthalate; Phe, phenanthrene.

could be the major causes of the observed cytotoxic and genotoxic effects induced by MNP exposure. Furthermore, MNPs can also affect the community structure of terrestrial plants. In a pot study with an artificially established plant community consisting of three grass species (*Festuca brevipila*, *Holcus lanatus*, and *Calamagrostis epigejos*) and four herbs (*Achillea millefolium*, *Hieracium pilosella*, *Plantago lanceolata*, and *Potentilla argentea*), Lozano and Rillig (2020) found that PET amendment (0.4% w/w) in soils promoted the dominance of *Calamagrostis epigejos* and *Hieracium pilosella*, but suppressed the growth of *Holcus lanatus* and *Festuca brevipila* by altering the soil properties. Given the foundational role of primary producers in terrestrial ecosystems, alterations in plant communities may have consequences for primary consumers and eventually for the structure of terrestrial food webs.

As effective sorbents, MNPs are known to concentrate organic chemicals or heavy metals from ambient media (Brennecke et al., 2016; Wang and Wang, 2018). The presence of MNPs may affect the availability of co-existing contaminants to terrestrial plants. In hydroponically grown lettuce (*Lactuca sativa* L.), the addition of MNPs to the culture solution was found to reduce the accumulation of di-*n*-butyl phthalate (DBP) in plant tissues (Gao et al., 2021b). Compared to treatment with arsenic (As) alone, lower plant uptake of As was observed in treatments containing MNPs (Dong et al., 2021; Dong et al., 2020b). This phenomenon was mainly attributed to the decreased aqueous concentrations of co-existing contaminants due to sorption by MNPs. However, it should be noted that most of these initial studies were conducted in laboratory settings, in which a fixed amount of co-present contaminants was spiked into the culture media only at the beginning of the exposure experiments. Sorption of co-occurring contaminants by MNPs will lead to a decrease in their bioavailable fraction for plant uptake. However, in realistic environmental scenarios, MNPs originating from sources such as biosolids are typically loaded with high concentrations of hazardous chemicals (Mohajerani and Karabatak, 2020). Once MNPs arrive at the rhizosphere zone, their interactions with root exudates may enhance the desorption of adhered chemicals from the contaminated MNPs. Additives leaching from MNPs during aging and weathering processes may contaminate soils in which terrestrial plants grow (Balestrini et al., 2014). Once MNPs are internalized into plants, the release of intrinsic and extrinsic contaminants from MNPs is also likely to occur under specific physiological conditions of plant tissues. Additionally, MNPs are potential carriers of bacteria and viruses, some of which can be pathogenic (Gkoutselis et al., 2021; Meng et al., 2021; Zettler et al., 2013). The potential role of MNPs in transporting the associated chemicals and pathogenic microbes to terrestrial plants requires further research.

3. Ingestion and impacts of MNPs in terrestrial fauna

3.1. Animal ingestion of MNPs

The environmental prevalence of MNPs renders these tiny polymeric particles readily available to a wide range of terrestrial animals. Although data regarding MNP ingestion in wild terrestrial animals are scarce, results from initial studies have demonstrated that this phenomenon does occur in the natural environment. In earthworm casts collected from traditional Mayan home gardens in southeast Mexico, MPs were detected at a concentration of 15 ± 29 items/g (Huerta Lwanga et al., 2017). Recently, the presence of MPs has been reported in the digestive systems of many other terrestrial invertebrates, including pillworm (*Porcellio scaber* Latreille), coridius chinensis (*Aspongopus chinensis* Dallas), dragonfly (*Anax parthenope Selys*), cricket (*Scapipedus aspersus* Walker), mole cricket (*Gryllotalpa africana* Palisot et Beaurois), locust (*Locusta migratoria* Linnaeus), eupolyphaga (*Eupolyphaga sinensis* Walker), Antarctic springtail (*Cryptopygus antarcticus*), slug (*Agriolimax agrestis*), and spider (*Araneus ventricosus* L. Koch) (Bergami et al., 2020; Lu et al., 2020). The occurrence of MNP ingestion has also been documented in some terrestrial vertebrates, including birds, herpetofauna, and mammals. Zhao et al. (2016) reported that 16 out of 17 terrestrial birds collected from Shanghai, China contained MPs in their digestive tracts. For birds of prey in central Florida, USA, MPs were observed in the gastrointestinal tracts of all the studied species, with an overall mean concentration of 11.9 ± 2.8 items/individual (Carlin et al., 2020). Using stomach flushing and fecal analysis, Mackenzie and Vladimirova (2021) investigated three species of terrestrial herpetofauna from southwestern Paraguay, including rococo toad (*Rhinella diptycha*), tropical house gecko (*Hemidactylus mabouia*), and Amazon lava lizard (*Tropidurus torquatus*), and found a total of 132 MPs in 81 out of 311 sampled individuals. The presence of MPs was also detected in the feces of ducks, geese, chickens, sheep, and pigs, indicating MPs have penetrated to poultry and livestock (Beriot et al., 2021; Huerta Lwanga et al., 2017; Reynolds and Ryan, 2018; Yang et al., 2021). Until recently, evidence of MP uptake by wild terrestrial animals has been acquired mainly from the analysis of animal feces and gastrointestinal contents. Little is known about the possible distribution of MPs in other tissues such as muscles, although very small particles (e.g., <130 μm) can penetrate the gut epithelial barriers through paracellular transport and disseminate to other organs via the circulatory system (Cox et al., 2019; Haave et al., 2021; Wright and Kelly, 2017). Additionally, owing to limitations in detection techniques, the uptake and accumulation of NPs in wild terrestrial animals are unknown.

In the laboratory, the uptake of MNPs has been documented in several terrestrial animal models, including earthworms, nematodes, snails, worms, silkworms, pillworms, mosquitoes, and rats (Table 3). Although after ingestion, the majority of MNPs were believed to be egested with feces, an appreciable portion of ingested MNPs remained within the animal body. Huerta Lwanga et al. (2016) exposed earthworms (*Lumbricus terrestris*) to PE MPs in litter and found that approximately 0.5% (w/w) of the ingested MPs was retained in earthworm guts. In a study of Wistar rats, Amereh et al. (2020) observed apparent accumulation of PS NPs in the testis region after oral exposure, suggesting the migration of NPs from the rat gut to other organs. Additionally, ontogenic transference of MPs was reported to occur in mosquitoes (*Culex pipiens*), which ingested MPs at the larval stage but still kept the MPs in their guts after they became adults (Al-Jaibachi et al., 2019). Long-term retention of MNPs in gastrointestinal systems or other organs of terrestrial animals may increase the possibility of trophic transfer of MNPs in the terrestrial food web.

Particle size has been demonstrated to be an important factor that can influence the ingestion of MNPs by terrestrial fauna (Al-Jaibachi et al., 2019; Huerta Lwanga et al., 2016; Lahive et al., 2019). Lahive et al. (2019) exposed terrestrial worms (*Enchytraeus crypticus*) to nylon particles of different sizes (13–150 μm) and found that the highest ingestion occurred for the smallest particles (13–18 μm). For earthworms (*Lumbricus terrestris*) exposed to MPs (<150 μm) in the litter, 90% of MPs in the casts was <50 μm, although the proportion of MPs within this size range in the original litter was only 50% (Huerta Lwanga et al., 2016). Terrestrial worms (*Enchytraeus crypticus*) and woodlice (*Porcellio scaber*) were also reported to preferentially take up shorter (12 μm–2.87 mm) MP fibers compared to the longer (4–24 mm) fibers (Selonen et al., 2020). These studies suggest that smaller MNPs are more likely to be ingested by some species. The concentrations of MNPs can affect their availability to exposed organisms. Significant positive correlations between the exposure concentrations of MNPs and MNP ingestion have been observed in some animals, including worms (*Enchytraeus crypticus*), woodlice (*Porcellio scaber*), and mosquito larvae (*Culex pipiens*) (Al-Jaibachi et al., 2019; Selonen et al., 2020). Additionally, it has been reported that the shape and color of MPs can affect the incidence of ingestion by some aquatic visual predators such as fish (Carlos de Sa et al., 2015; Wang et al., 2020b). Similar scenarios may also occur among terrestrial visual predators. The formation of biofilms on MNPs may produce specific gustatory or olfactory cues, which may lead to increased active ingestion of MNPs by organisms with chemoreceptors (Carbery et al., 2018). Moreover, aspects of terrestrial fauna such as species, growth stage, trophic level, and food strategy may also impact the incidence of MNP ingestion. Further research is required to clarify to what extent and how these factors affect the uptake of MNPs in terrestrial fauna.

3.2. Effects of MNPs on animal health

Exposure to MNPs has been shown to cause a variety of physical impacts on terrestrial fauna (Table 3). Once ingested, MNPs can induce histopathological damage to the gastrointestinal tracts of terrestrial animals. In a study with snails (*Achatina fulica*), Song et al. (2019) observed serious damage to the villi in the gastrointestinal walls of test organisms after 28 days of PET MP exposure. Similar effects were reported in nematodes (*Caenorhabditis elegans*) and earthworms (*Eisenia andrei*) (Rodriguez-Seijo et al., 2017; Yu et al., 2020b). Ingestion of MNPs has also been reported to cause alterations in the intestinal microflora of animals such as earthworms (*Metaphire californica*), springtails (*Folsomia candida*), worms (*Enchytraeus crypticus*), honey bees (*Apis mellifera* L.) and some mammals (Banerjee and Shelver, 2021; Cheng et al., 2021; Wang et al., 2019a; Wang et al., 2021b; Zhu et al., 2018). Accumulation of MNPs in the gut can diminish the feeding impetus of animals, which may eventually result in reduced growth and development (Huerta Lwanga et al., 2016; Selonen et al., 2020; Song et al., 2019). Oxidative stress is another common symptom induced by MNP exposure. Yu et al. (2020b) found a concentration-dependent increase in ROS production in nematodes (*Caenorhabditis elegans*) under exposure to PS MPs. In earthworms (*Eisenia fetida*) exposed to PE films, significant decreases were observed in the activities of antioxidant (e.g., SOD and CAT) and detoxifying (e.g., glutathione-*S*-transferase) enzymes, suggesting that MNP exposure could impair the antioxidant system (Cheng et al., 2020). MNPs can also induce genotoxicity by altering the expression of antioxidant- and immunity-related genes, making the exposed animals highly susceptible to biotic or abiotic stresses (Cheng et al., 2020; Muhammad et al., 2021). Additionally, ingested MNPs may migrate from the animal gut to other organs, causing tissue impairment, endocrine disruption, reproductive toxicity, and neurotoxicity (Amereh et al., 2020; Banerjee and Shelver, 2021). In cases where animals are exposed to high concentrations of MNPs, increased mortality may occur (Huerta Lwanga et al., 2016; Ju et al., 2019). However, there have been some studies that reported negligible effects of MNPs on terrestrial fauna. For instance, Rodriguez-Seijo et al. (2017) found that exposure to MPs exerted no significant effects on survival, weight, and juvenile number of earthworms (*Eisenia andrei*). In another study, negligible changes in biochemical parameters were observed in earthworms (*Eisenia fetida*) after exposure to MPs (<10% w/w in soil). Given the inconsistency of reported results, additional studies are urgently needed to unveil the mechanisms underlying the effects of MNPs, especially at the environmentally relevant concentrations, on terrestrial fauna. Furthermore, since most of the previous toxicity studies are based on a single species for testing (Table 3), the effects of MNPs on faunal populations and communities and the associated implications for

Table 3
Ingestion and biological effects of MNPs in terrestrial animals.

Animal species	MNPs			Co-present contaminants	Observations	Reference
	Polymer type	Particle size	Concentration			
MNPs alone						
Earthworm (*Lumbricus terrestris*)	PE	<150 μm	7–60% (w/w in litter)	/	Ingestion and egestion of PE; Increase in mortality; Reduced growth rate.	(Huerta Lwanga et al., 2016)
Nematode (*Caenorhabditis elegans*)	PS	1 μm	1–100 μg/L	/	Ingestion of PS; Increase in ROS production; Intestinal damage.	(Yu et al., 2020b)
Springtail (*Folsomia candida*)	PE	<500 μm	0.1–1% (w/w in dry soil)	/	Avoidance behavior; Decrease in reproduction; Changes in gut microbial community; Mortality.	(Ju et al., 2019)
Snail (*Achatina fulica*)	PET	1.3 mm	0.01–0.71 g/kg soil	/	Ingestion and egestion of PET; Reduced food intake and excretion; Decrease in total antioxidant capacity in liver; Intestinal villi damage.	(Song et al., 2019)
Worm (*Enchytraeus crypticus*)	PA	13–150 μm	2–12% (w/w in soil)	/	Ingestion of PA; Reduced reproduction.	(Lahive et al., 2019)
Silkworm (*Bombyx mori*)	PS	0.05–6 μm	10 μg/L PS solution sprayed on mulberry leaves.	/	Accumulation of PS in gut tissue and lumen; Alterations in gene expression; Oxidative stress; Reduced immunity to pathogens.	(Muhammad et al., 2021)
Woodlice (*Porcellio scaber*)	PET	10–3000 μm	0.02–1.5% (w/w in soil)	/	Ingestion of PET; Reductions in energy reserve and feeding activity.	(Selonen et al., 2020)
Mosquito (*Culex pipiens*)	PS	2–15 μm	50–200 particles/mL	/	Ingestion of PS; Ontogenic transference of PS; No impact on the growth or mortality.	(Al-Jaibachi et al., 2019)
Honey bees (*Apis mellifera* L.)	PS	25 μm	0.5–10 mg/L	/	Ingestion of PS; Alterations in gene expression; Changes in gut microbiome.	(Wang et al., 2021b)
Rat	PS	40 nm	1–10 mg/kg body weight day	/	Uptake of PS; High accumulation of PS in testis; Endocrine disruption; Tissue and cell impairment; Reproductive toxicity; Alterations in gene expression.	(Amereh et al., 2020)
MNPs with other contaminants						
Earthworm (*Metaphire californica*)	PVC	/	2 g/kg	As (40 mg/kg)	Decrease in bioaccumulation of total As; Alleviation of As toxicity to gut microbiome.	(Wang et al., 2019a)
Earthworm (*Eisenia fetida* & *Metaphire guillelmi*)	PS	<2 mm	0.25% (w/w in soil)	HBCDDs (~40 μg/kg)	Ingestion of PS; Increased bioaccumulation of HBCDDs;	(Li et al., 2019)
Earthworm (*E. fetida*)	PE	30–100 μm	0.1–10% (*w*/w in soil)	Ni^{2+} (40 mg/kg) & Cu^{2+} (100 mg/kg)	Increased bioaccumulation of metals; Oxidative stress; Alterations in gene expression.	(Li et al., 2021)
Earthworm (*E. fetida*)	LDPE & PS	<300 μm	0.1–10% (w/w in soil)	PAHs (510–740 μg/kg) & PCBs (180–220 μg/kg)	Ingestion of LDPE and PS; Oxidative stress; Decreased bioaccumulation of PAHs and PCBs	(Wang et al., 2019b)
Worm (*E. crypticus*)	PS	50–100 nm	1 g/kg in oats	Tetracycline (10 mg/kg)	Ingestion of PS; Increased bioaccumulation of tetracycline; Increase in the diversity and abundance of ARGs in gut microbiome; Gut microbiome dysbiosis.	(Ma et al., 2020)
Woodlice (*P. scaber*)	PET	10–3000 μm	0.5% (w/w in soil)	Chlorpyrifos (0.2–2 mg/kg)	Decreased bioaccumulation of chlorpyrifos; Increase in total hemocyte count; Slight alterations in immune processes.	(Dolar et al., 2021)

PE, polyethylene; PS, polystyrene; PET, polyester; PA, polyamide; PVC, polyvinylchloride; LDPE, low-density polyethylene; As, arsenic; HBCDDs, hexabromocyclododecanes; Ni, nickel; Cu, copper; PAHs, polycyclic aromatic hydrocarbons; PCBs, polychlorinated biphenyls; ROS, reactive oxygen species; ARGs, antibiotic resistance genes.

the structure and stability of the terrestrial food web remain completely unknown.

MNPs acquired from the environment typically contain high levels of hazardous chemicals (e.g., plasticizers, heavy metals, and hydrophobic organic contaminants) and can serve as strong sinks for these chemicals (Holmes et al., 2012; Kwon et al., 2017). Besides the physical toxicity caused by MNPs per se, the co-present hazardous chemicals may cause additional effects to the terrestrial animals after being ingested along with MNPs (Table 3). Some studies have demonstrated that MNP ingestion can increase the bioavailability of the associated chemicals. Li et al. (2019) exposed earthworms (*Eisenia fetida* and *Metaphire guillelmi*) to hexabromocyclododecanes (HBCDDs) by amending the soil with PS MPs that contained HBCDDs as additives or artificially contaminating the soil with a similar level of HBCDDs. They found that the concentrations of HBCDDs in earthworms from treatments with PS MPs were much higher than those from the MP-free treatments and attributed it to the enhanced release of HBCDDs from the ingested MPs under the specific conditions of earthworm digestive fluid. It was also reported that ingestion of PS NPs could facilitate bioaccumulation of tetracycline in worms (*Enchytraeus crypticus*), leading to increased diversity and abundance of antibiotic resistance genes (ARGs) and microbiome dysbiosis in the gut (Ma et al., 2020). Additionally, the incorporation of MNPs in soils could reduce the sorption of heavy metals (e.g., copper and nickel) by soils, making them readily available to earthworms (*Eisenia fetida*) (Li et al., 2021). However, MNPs do not always enhance the bioaccumulation and toxicity of co-present contaminants in terrestrial animals. In earthworms (*Metaphire californica*), the presence of PVC MPs reduced the bioaccumulation of total arsenic and alleviated arsenic toxicity to the gut microbiome (Wang et al., 2019a). Similarly, a reduction in chlorpyrifos intake was observed in woodlice (*Porcellio scaber*) from treatments with PET MPs (Dolar et al., 2021). Although contrasting, results from these preliminary studies suggest that MNPs can play a role in modulating the bioavailability and toxicity of co-existing pollutants to terrestrial fauna. Furthermore, MNPs are also carriers of microbial contaminants (e.g., bacterial pathogens and ARGs) and may affect the composition of gut microbiome once ingested by terrestrial animals.

4. Trophic transfer of MNPs

Although scarce, some preliminary studies have demonstrated that MNPs can transfer across trophic levels in terrestrial food webs. Huerta Lwanga et al. (2017) observed a stepwise increase in the concentrations of MPs from home garden soil (0.87 ± 1.9 particles/g) to earthworm casts (14.8 ± 28.8 particles/g) and chicken feces (129.8 ± 82.3 particles/g) and proposed that trophic transfer of MPs might occur in this short terrestrial food chain. Additionally, MPs have been detected in the gastrointestinal tracts or feces of field-collected terrestrial birds of prey, including red-shouldered hawk (*Buteo lineatus*), barred owl (*Strix varia*), eastern screech owl (*Megascops asio*), black vulture (*Coragyps atratus*), red-tailed hawk (*Bueto jamaicensis*), and coopers hawk (*Accipiter cooperii*) (Carlin et al., 2020). Some herpetofauna, such as rococo toad (*Rhinella diptycha*), tropical house gecko (*Hemidactylus mabouia*), and Amazon lava lizard (*Tropidurus torquatus*), from Southwestern Paraguay were also reported to contain MPs (Mackenzie and Vladimirova, 2021). These indicate the incidence of trophic transfer of MPs in the terrestrial wildlife. In a microcosmic study, Chae and An (2020) found that PS NPs (20 nm) could be internalized in mung bean (*Vigna radiata*) via root uptake and then transferred to the African giant snail (*Achatina fulica*) that fed on mung bean leaves. This provides an initial clue for the trophic transfer of MNPs from terrestrial primary producers to low-trophic consumers.

In terrestrial ecosystems, plants can create organic matter via photosynthesis using energy from sunlight, which forms the basis of the terrestrial food web. Internalization of MNPs in plants via root or foliar uptake has been demonstrated, indicating that MNPs are capable of penetrating the producers of the terrestrial food web (Li et al., 2020a; Sun et al., 2021). Trophic transfer of MNPs can occur when plant structures containing MNPs are consumed by primary consumers (Chae and An, 2020). Although the ingested MNPs are mainly retained in animal gastrointestinal tracts or excreted with feces, there is evidence of translocation of MNPs from the gut to other organs (Amereh et al., 2020; Banerjee and Shelver, 2021). Additionally, some animals, such as small soil-dwelling invertebrates, are usually ingested as a whole by their predators. In either case, predator–prey interactions can lead to the transfer of MNPs up the trophic levels. However, there remain large knowledge gaps regarding the bioaccumulation and biomagnification potentials of MNPs in the terrestrial food web. The potential role of MNPs in the transfer of associated contaminants along the terrestrial food chain is poorly understood and deserves further research.

5. Intake and impacts of MNPs in humans

5.1. Dietary intake of MNPs

MNPs have been detected in a variety of foods and beverages, including seafood, edible crops, table salts, canned fish, take-out food, honey, beer, milk, and drinking water (Table 4). Seafood represents a substantial source of MNPs for human ingestion. For instance, the number of MPs in commercial marine bivalves from China was reported to be 4.3–57.2 items/individual, which may lead to an annual intake of 100,000 MPs per capita in Chinese shellfish consumers (Li et al., 2015). Other marine species such as fishes, shrimps, and crabs were also reported to contain MPs, although little is known about the presence of MPs in the edible parts of these species (Kwon et al., 2020). It is estimated that >150 MPs are ingested daily by the Japanese owing to the consumption of seafood (Cox et al., 2019). MNPs have been detected in some market-sold edible crops from Catania, Italy, including carrot (*Daucus carota*), lettuce (*Lactuca sativa*), broccoli (*Brassica oleracea italica*), potato (*Solanum tuberosum*), apple (*Malus domestica*), and European pear (*Pyrus communis*), which may lead to the dietary intake of MNPs in local consumers (Oliveri Conti et al., 2020). An average of 10.2 ± 13.8 MPs/gizzard was detected in chicken gizzards intended for human consumption (Huerta Lwanga et al., 2017). Processed foods and beverages are also frequently reported to be contaminated with MNPs (Table 4), although the contamination is more likely to originate from the production or preparation processes rather than from the environment. For instance, steeping a plastic teabag in hot water can release a considerable number of MNPs into the beverage (Hernandez et al., 2019). A person who orders take-out food 4–7 times per week may ingest 12–200 MPs flaked from plastic food-contacting materials (Du et al., 2020). When ~15% of the caloric intake was evaluated, the annual dietary intake of MPs via food and beverage in Americans was estimated to range from 39,000 to 52,000 particles per capita depending on age and sex (Cox et al., 2019).

Direct evidence of MNP intake by humans is scarce and primarily comes from the analysis of human feces. Schwabl et al. (2019) investigated the presence of MPs in human feces and found that all stool samples collected from eight participants varying in age, gender, and nationality contained MPs (50–500 μm) with a median concentration of 20 items/10 g. PET and polycarbonate (PC) MPs were detected in the feces of newborns (PET: 0–12,000 ng/g dry weight (dw); PC: 0–110 ng/g dw), 1-year-old infants (PET: 5700–82,000 ng/g dw; PC: 49–2100 ng/g dw), and adults (PET: 0–16,000 ng/g dw; PC: 37–620 ng/g dw) from New York State, USA, which were considered to be mainly derived from dietary sources (Zhang et al., 2021a). The extensive use of plastic products such as baby feeding bottles, utensils, sippy cups, and food containers and frequent sucking or touching of textiles such as clothes and carpets were assumed as the possible reasons for the higher MP intake in infants than in adults. The presence of MPs was also reported in the feces of Chinese people from different cities, and participants who preferred take-out food and bottled beverages tended to excrete higher concentrations of MPs (Yan et al., 2021; Zhang et al., 2021b). These preliminary studies suggest that dietary intake is a notable pathway for MPs to enter the human body. Although human ingestion of NPs may also occur, until recently, there have been no quantitative data to support this.

5.2. Effects of MNPs on human health

Owing to ethical constraints and safety issues, there have been no in vivo studies on the impacts of MNPs on human health. However, a good number of studies have verified the negative health impacts associated with MNP exposure using mammalian models, such as mice. Deng et al. (2017) found that after ingestion, PS MPs (5–20 μm) could accumulate in gut, liver and kidney of mice (*Mus musculus*), causing oxidative stress, energy deficiency, lipid disturbance, and neurotoxicity. Ingested PS NPs (40 nm) could also translocate to the testes of male Wistar rats, leading to serious histological impairments, lower concentrations and quality of serums, and significant alterations in reproductive hormones in testis tissues (Amereh et al., 2020). This indicates that MNP exposure can induce endocrine disturbance and reproductive toxicity in male rats. Additionally, maternal exposure to PS MNPs during gestation and lactation periods of mice (*Mus musculus*) was found to cause metabolic disorders in both the dams and their offsprings, suggesting the intergenerational effects of MNPs

Table 4
The presence of MNPs in commonly consumed foods and beverages.

Food/beverage	Location	MNPs			Reference
		Size range	Polymer type	Abundance	
Commercial bivalves	China	5 μm–5 mm	PE, PET, & PA	2.1–10.5 items/g	(Li et al., 2015)
Canned sardines & sprats	International, 13 countries	0.2–3.8 mm	PP, PET, PE, & PVC	A total of 6 items in 4 out of 16 brands.	(Karami et al., 2018)
Chicken gizzard	Mexico	0.1–5 mm	/	10.2 ± 13.8 items/individual	(Huerta Lwanga et al., 2017)
Carrot, lettuce, broccoli, potato, apple, & pear	Italy	1.5–2.5 μm	/	26,000–310,000 items/g	(Oliveri Conti et al., 2020)
Take-out food	China	0.04–5 mm	Mainly, PS, PE, PET, & PP	3–29 items/container	(Du et al., 2020)
Edible salts	International, 17 countries/regions	0.1–5 mm	Mainly, PE, PP, & PET	Sea salt: 0–13,629 items/kg; Rock salt: 0–148 items/kg; Lake salt: 28–462 items/kg.	(Kim et al., 2018)
Honey	International, 9 countries	>40 μm	/	Fiber: 10–336 items/kg; Fragment: 2–82 items/kg	(Liebezeit and Liebezeit, 2015)
Beer	America	0.1–5 mm	/	0–14.3 items/L	(Kosuth et al., 2018)
Milk	Mexico	0.1–5 mm	PES & PSU	3–11 items/L	(Kutralam-Muniasamy et al., 2020)
Tap water	International, 14 countries	0.1–5 mm	/	0–61 items/L	(Kosuth et al., 2018)

PE, polyethylene; PET, polyester; PS, polystyrene; PA, polyamide; PVC, polyvinylchloride; PES, polyethersulfone; PSU, polysulfone.

(Luo et al., 2019). Deng et al. (2018) investigated the effect of MPs on toxicity of two organophosphorus flame retardants [i.e., tris (2-chloroethy) phosphate (TCEP) and tris (1,3-dichloro-2-propyl) phosphate (TDCPP)] in mice (*Mus musculus*) and found that MPs significantly aggravated the toxicity of co-present chemicals. In another study, co-exposure with MPs was observed to alleviate the toxic effects of tributyltin on mice (*Mus musculus*) (Jiang et al., 2021). These highlight the controversial role of MNPs in toxicity of co-present toxicants. It remains unveiled whether the observed health risks of MNPs to model animals are applicable to human beings, which needs further clarification.

Available in vitro studies using established human-derived cell lines have demonstrated that MNPs can pass through the cytomembranes into cells of various types (Table 5). Cortés et al. (2020) investigated the uptake of PS NPs (50–100 nm) by human intestinal epithelial cells (Caco-2) and found that >80% of the test cells contained PS particles after exposure to 100 μg/mL of PS NPs for 24 h. Cellular internalization of MNPs has also been observed in other human cell lines, including lymphoblast cells, alveolar epithelial cells, and hepatic cells (Domenech et al., 2020; He et al., 2020; Xu et al., 2019). Endocytosis is considered the key mechanism for the cellular internalization of MNPs (Yameen et al., 2014; Wang et al., 2020a). The potential of MNPs to be internalized in these model cell lines indicates that MNPs may migrate to different organs after entering the human body.

The reported toxicological effects of MNPs on human cell lines mainly include oxidative stress, reduced viability, inflammation, genotoxicity, and apoptosis (Table 5). Exposure to MNPs can cause ROS generation and impairment of antioxidant systems, even in cases where MNPs are not internalized in the test cells (Dong et al., 2020a; He et al., 2020; Schirinzi et al., 2017). For instance, He et al. (2020) found that 100 μg/mL of PS-NH_2 caused the maximum accumulation of malondialdehyde in human hepatic cells (HepG2), accompanied by a decrease in SOD activity and glutathione content. A concentration-dependent decrease in cell viability was observed in lung epithelial cells (BEAS-2B) after exposure to 10–1000 μg/cm^2 PS MPs (Dong et al., 2020a). MNP-associated reductions in cell viability have also been documented in other cell models, including hepatic cells (HepG2), intestinal epithelial cells (Caco-2), peripheral blood mononuclear cells (PBMCs), mast cells (HMC-1), and dermal fibroblasts (HDFs) (He et al., 2020; Huang et al., 2021; Hwang et al., 2019). Studies on human alveolar epithelial cells (BEAS-2B and A549) have shown that MNPs can induce cellular inflammatory responses by altering the transcription levels of relevant genes (Dong et al., 2020a; Xu et al., 2019). Inflammation reactions can destroy the integrity of the cell membrane, causing cell necrosis and finally leading to the development of various diseases. Cell apoptosis may also occur due to the upregulated expression of apoptotic proteins following MNP exposure (Xu et al., 2019). However, the specific effects of MNPs on human cells can differ significantly between studies. Domenech et al. (2020) found that although PS NPs (50–100 nm) were internalized by human colorectal cells (Caco-2 and HT-29-MTX) and lymphoblast cells (Raji-B), no significant adverse effects on cell viability, membrane integrity, ROS production, DNA structure, and gene expression were observed. Similar results were also reported by Cortés et al. (2020), who observed a clear internalization of PS NPs but slight cytotoxic/genotoxic effects in the test cells. The inconsistency in toxicity results calls for further studies to acquire a comprehensive and mechanistic understanding of the impact of MNPs on human cell lines.

Many factors can affect the uptake and toxicity of MNPs in human cells. Compared to large MNPs, smaller particles tend to be more easily taken up by human cells. This has been well demonstrated in a recent study in which the uptake of PS MNPs of varying sizes (i.e., 0.3, 0.5, 1, 3, and 6 μm) in human colonic cells (Caco-2) was 73%, 71%, 49%, 43%, and 30%, respectively (Wang et al., 2020a). Similarly, Xu et al. (2019) also found a more efficient uptake of small PS NPs (25 nm) in human lung epithelial cells (A549) compared to that of larger particles (70 nm). The enhanced uptake of smaller MNPs may be partially responsible for their higher cytotoxic effects (Xu et al., 2019). Cortés et al. (2020) reported that the cellular uptake of PS NPs (1–100 μg/mL) was concentration-dependent. Some MNP-induced intracellular responses, such as decreased cell viability, oxidative stress, inflammation, and genotoxicity, were generally found to be more significant at higher exposure concentrations (Dong et al., 2020a; He et al., 2020; Hwang et al., 2019; Xu et al., 2019). In comparison to the unmodified PS NPs, PS-COOH and PS-NH_2 exhibited higher internalization in hepatic cells (HepG2), indicating specific interactions of the carboxyl/amino groups with receptors on the cell surface might occur, which facilitated the endocytosis of surface-functionalized particles (He et al., 2020). The resultant cytotoxicity was also found to differ considerably between treatments, with more serious effects observed under exposure to PS-NH_2 (He et al., 2020). For cerebral cells (T98G) and cervical epithelial cells (HeLa) exposed to PE MNPs, only T98G cells showed significant ROS generation, while PS-treated cells both generated higher levels of ROS (Schirinzi et al., 2017). This implies that the specific cytotoxic effects of MNPs are influenced by both the polymer type and cell line (Schirinzi et al., 2017; Shi et al., 2021a; Stock et al., 2021). To summarize, particle size, exposure concentration, surface chemistry, and polymer types of MNPs and cell types are potential factors that influence the internalization and toxicology of MNPs in human cell lines.

Although highly limited, some initial in vitro studies have demonstrated that co-exposure to MNPs can affect the toxicity of other contaminants to human cells (Table 5). Huang et al. (2021) investigated the

Table 5
In vitro studies on the impacts of MNPs on different human cell lines.

Cell lines	MNPs			Co-present contaminants	Observations	Reference
	Polymer type	Particle size	Concentration			
MNPs alone						
Intestinal epithelial cells (Caco-2)	PS	50–100 nm	25–200 μg/mL	/	Cellular internalization of PS; Slight cytotoxic/genotoxic effects.	(Cortés et al., 2020)
Colorectal cells (Caco-2 & HT-29-MTX) & lymphoblast cells (Raji-B)	PS	50–100 nm	1–200 μg/mL	/	Cellular internalization of PS; No significant effects on cell viability, membrane integrity, ROS production, DNA, and gene expression.	(Domenech et al., 2020)
Lung epithelial cells (BEAS-2B)	PS	1.7–2.2 μm	1–1000 μg/cm^2	/	Reduction in cell viability; Alterations in cell morphology; Increased ROS production; Cellular inflammatory response.	(Dong et al., 2020a)
Alveolar epithelial cells (A549)	PS	25–70 nm	1.1–25 μg/mL	/	Cellular internalization of PS; Decrease in cell viability under high exposure concentrations; Cellular inflammatory response; Disturbance in cell cycle and apoptosis; Alterations in gene and protein expressions.	(Xu et al., 2019)
Hepatic cells (HepG2)	PS, PS-COOH, & PS-NH_2	50 nm	10–100 μg/mL	/	Cellular internalization of NPs; Decrease in cell viability; Cellular oxidative stress.	(He et al., 2020)
Cerebral cells (T98G) & cervical epithelial cells (HeLa)	PE & PS	PE: 0.1–16 μm PS: 0.04–10 μm	0.01–10 μg/mL	/	No significant effects on cell viability; Cellular generation of ROS in PE-treated T98G cells and PS treatments.	(Schirinzi et al., 2017)
Peripheral blood mononuclear cells (PBMCs), mast cells (HMC-1), & dermal fibroblasts (HDFs)	PP	20–200 μm	10–1000 μg/mL	/	No cytotoxicity for PP of >25 μm; Decreased cell viability and increased ROS production under exposure to 1000 μg/mL of ~20 μm PP.	(Hwang et al., 2019)
MNPs with other contaminants						
Intestinal epithelial cells (Caco-2)	PE	30–140 μm	100–1000 μg/mL	TBBPA (10–50 mg/L)	Decrease in cell viability; Increased ROS generation; Reduction in mitochondrial membrane potential; Increased release of lactate dehydrogenase; Joint toxicity of PE with TBBPA to a certain degree.	(Huang et al., 2021)
Intestinal epithelial cells (Caco-2)	PS	0.3–6 μm	20–120 μg/mL	BPA (20–120 ng/mg)	Cellular internalization of PS; Reduced cell viability; Increase in intracellular ROS production; Mitochondrial depolarization; Synergistic toxicity of PS with BPA.	(Wang et al., 2020a)
Alveolar epithelial cells (A549)	PS	100 nm	10–1000 μg/mL	DBP & DEHP (5 μg/mL)	Cellular internalization of PS; Increase in cytotoxicity of DBP and DEHP under higher concentrations of PS; Decreased bioavailability of DBP and DEHP.	(Shi et al., 2021b)

PS, polystyrene; PE, polyethylene; PP, polypropylene; TBBPA, tetrabromobisphenol A; BPA, bisphenol A; DBP, dibutyl phthalate; DEHP, di-(2-ethylhexyl)phthalate; ROS, reactive oxygen species.

effects of PE MPs and tetrabromobisphenol A (TBBPA) on human intestinal epithelial cells (Caco-2) and found that both PE MPs and TBBPA could cause detrimental effects in Caco-2 cells and that PE MPs exhibited slight joint toxicity with TBBPA. Another study with Caco-2 cells also found that exposure to PS NPs that had been loaded with bisphenol A (BPA) via sorption caused reduced cell viability, higher generation of intracellular ROS, and mitochondrial depolarization, suggesting the synergetic cytotoxic effects of PS NPs and BPA. However, the toxicity results of MNPs and co-present contaminants are not always consistent among studies. Shi et al. (2021b) observed that, when co-exposed, low concentrations of PS NPs (20 μg/mL) could alleviate the toxicity of two phthalate esters (PAEs) [dibutyl phthalate (DBP) and di-(2-ethyl hexyl) phthalate (DEHP)] to human alveolar epithelial cells (A549), probably resulting from the reduced dissolved concentrations of PAEs due to sorption by PS NPs. However, at higher NP concentrations (200 μg/mL), more toxic effects were observed compared to that of single PAE exposure, which was attributed to the dominant role of PS NPs in the combined cytotoxicity (Shi et al., 2021b). Therefore, the potential role of MNPs in the bioavailability of co-present chemicals in human cells remains debatable and deserves further study.

6. Conclusions and prospects

The prevalence of MNPs in terrestrial ecosystems leads to inevitable encounters and interactions between terrestrial biota and these small polymeric particles. Research on the uptake and impact of MNPs in terrestrial organisms is still in its infancy but has got increasing momenta in recent years. This review summarizes the current research progress regarding the uptake, trophic transfer, and toxicological impacts of MNPs in the terrestrial food web from plants to humans. The uptake of MNPs has been documented for a wide range of terrestrial species representing different trophic levels. Owing to limitations in detection techniques, field data regarding the uptake and fate of NPs in terrestrial biota are currently very scarce. The observed biological effects associated with MNP exposure are diverse and often contradictory, which seem to be influenced by the particle properties, test species, and experimental conditions. However, the environmental relevance of some laboratory toxicity studies is in debate. The underlying mechanisms of MNP-induced toxicity remain unclear. The effects of MNPs on the bioaccumulation of co-present contaminants can be contradictory between studies. Limited studies have been conducted to evaluate the biological effects of MNPs at the population or community level and the resultant implications for the terrestrial food web. Dietary intake represents a primary way for MNPs to enter the human body. However, quantitative data on MNP intake in humans via consumption of foods, especially the edible parts of terrestrial plants and animals, are still limited. It remains unclear to what extent of MNP contamination in food items is owing to the bioaccumulation of MNPs in edible plant or animal tissues. To achieve a more comprehensive and systematic understanding of the ecotoxicological effects of MNPs on the terrestrial food web, several research priorities are recommended as follows:

(1) Field data regarding the presence of MNPs in edible plant tissues and eviscerated flesh of animals are scarce. Large-scale field investigations are needed to quantify the occurrence of MNPs in different tissues of a broader suite of terrestrial species representing different trophic levels.
(2) Pristine and spherically shaped PS MNPs have been the most widely used test materials in previous laboratory studies. Whereas, in the natural environment, multi-polymeric, irregularly shaped (e.g., fibrous and fragmental), partially degraded, and bio-fouled particles are the most ubiquitous. Future studies should consider the effects of factors such as polymer type, shape, biofouling, and aging of MNPs on their uptake and toxicity in terrestrial organisms. The potential role of MNPs as vectors to transfer biofilm-associated pathogenic bacteria, fungi, and viruses into terrestrial biota also deserves attention.
(3) Most of the previous studies have used a single species to test MNP effects. The effects of MNPs on the community structure of terrestrial biota and their associated implications for the terrestrial food web require further clarification by encompassing multi-species assemblages in future research.
(4) Many studies have focused on the short-term biological effects of MNPs, which usually employ high exposure concentrations. Further research is needed to reveal the chronic impacts of MNPs on terrestrial species following long-term exposure to MNPs at environmentally relevant concentrations.
(5) The potential role of MNPs in transferring associated contaminants to terrestrial biota remains under debate, which deserves further elaborate studies for clarification. Special attention should be paid to NPs that are more sorptive to extrinsic chemicals because of their increased surface areas and are more effective at passing through biological membranes because of their smaller size than the larger-sized MPs. Additionally, whether the specific physiological conditions in biological matrices could enhance the release of intrinsic chemicals (e.g., additives) from the internalized MNPs also requires further research.
(6) Further research is needed to demonstrate the trophic transfer of MNPs and their internalization via food consumption as well as the associated contaminants in the terrestrial food web.
(7) As for food contamination by MNPs, future work needs to quantify the occurrence of MNPs in food items, such as grains, vegetables, fruits, livestock, and poultry, which represent the main sources of nutrition for humans worldwide.
(8) Based on the reported data, the estimated dietary intake of MNPs in humans is far greater than that detected in human feces, which raises a question about the fate of the remaining MNPs in human bodies. Therefore, further studies are needed to identify the fate and health effects of internalized MNPs in human bodies by using mammalian and cellular models.

CRediT authorship contribution statement

Wenfeng Wang: Conceptualization, Investigation, Formal analysis, Writing – original draft. **Anh T. Ngoc Do:** Conceptualization, Investigation, Writing – review & editing. **Jung-Hwan Kwon:** Conceptualization, Supervision, 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.

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