CellPress

Trends in Plant Science

Special issue: Root biology and soil health for a sustainable future

Review

Ecological effects of micro/nanoplastics on plant-associated food webs

Muhammad Ilyas[1,2], Carlos M. Duarte[3], Elvis Genbo Xu[4], Guorui Xu[1,*], and Jie Yang[1,*]

Micro/nanoplastics (MNPs) contamination is a potential threat to global biodiversity and ecosystem functions, with unclear ecological impacts on aboveground (AG) and belowground (BG) food webs in terrestrial ecosystems. Here, we discuss the uptake, ingestion, bioaccumulation, and ecotoxicological effects of MNPs in plants and associated AG-BG biota at various trophic levels. We propose key pathways for MNPs transfer between the AG-BG food webs and elaborate their impact on terrestrial ecosystem multifunctionality. We conclude that MNPs are bioaccumulated in most studied plants and associated AG-BG biota and can be transferred along AG-BG food webs, which may profoundly impact ecosystem functioning. However, most pathways are still untested. Future research on MNPs should focus on the interactions within AG-BG food webs in terrestrial ecosystems.

Highlights

Micro/nanoplastics (MNPs) are ubiquitous across environments and are becoming an emerging threat to both marine and terrestrial ecosystems.

Although the ecological impacts of MNPs on marine ecosystems have already received considerable attention, their effects on terrestrial biota at various trophic levels and their transfer through the aboveground–belowground (AB-BG) food webs are still far from clear.

Most studies show that MNPs bioaccumulate in plants and AG-BG biota, leading to a range of adverse ecotoxicological effects at various trophic levels.

More studies are needed on long-term effects of MNPs on AG-BG interactions, biodiversity, and ecosystem functions.

MNPs in terrestrial ecosystems

The ever-growing release of plastic waste into the environment adds to the existing environmental crises [1], potentially impacting both the environment and human health [2,3]. The global gross production of virgin plastics was estimated to be 8300 million tonnes (Mt; as of 2015), with a compound annual growth rate of 8.4%, outweighing most other man-made materials since 1950 [4]. Approximately 1.2 billion tonnes of plastic waste have been discarded or accumulated in landfills or the natural environment [4]. Most plastic waste is recalcitrant and persists in the environment [5] but may be broken up into smaller pieces by biotic and abiotic weathering [6]. Plastic pieces smaller than 5 mm are defined as **MNPs** (see Glossary) [7].

MNPs are ubiquitous across various environments, ranging from deserts to forests, mountain peaks to the deep ocean, tropical landfills, and Arctic snow [8,9]. Most assessments of the impact of MNPs have primarily focused on marine life [10], leading the United Nations to agree to establish a legally binding mechanism to prevent plastics from entering the marine environment by 2024 [2]. However, MNPs were recently identified as emerging threats to terrestrial ecosystems [11]. Sources of MNPs in terrestrial ecosystems are primarily linked to industry, agriculture, and atmospheric deposition (Figure S1 in the supplemental information online). The reported abundance (0.34 to 410.95 particles/kg) and concentration (0.002 to 67 500 mg/kg) of soil MNPs is highly variable across sites [12]. Over 8.9 Mt of MNPs leak into land, and 3.8 Mt into oceans annually [13], indicating a total leakage of 10–40 Mt, with land leakage being 3–10 times higher than that of oceans [14]. Research on terrestrial MNPs lagged two decades behind marine studies (Figure S2 in the supplemental information online).

Terrestrial MNPs may change the soil physicochemical properties, affecting element cycling and greenhouse gas emissions [15]. One possible mechanism underlying the effects of MNPs on **ecosystem multifunctionality** is that soil microorganisms metabolise MNPs and their associated leachates, driving shifts in key biogeochemical cycles [16]. For example, MNPs alter soil structure and impact pore space, water retention, aeration, and microbial activity, thereby affecting carbon and nitrogen cycles

[1]State Key Laboratory of Plant Diversity and Specialty Crops, Xishuangbanna Tropical Botanical Garden, Chinese Academy of Sciences, Menglun, Yunnan, 666303, China
[2]University of Chinese Academy of Sciences, Beijing 100049, China
[3]Marine Science Program, Biological and Environmental Science and Engineering Division, King Abdullah University of Science and Technology (KAUST), Thuwal 23955-6900, Kingdom of Saudi Arabia
[4]Department of Biology, University of Southern Denmark, Odense 5230, Denmark

*Correspondence:
xuguorui@xtbg.ac.cn (G. Xu) and yangjie@xtbg.org.cn (J. Yang).

[16]. Likewise, MNPs may affect phosphorus cycling by promoting sulfate reduction, likely facilitated by *Desulfovibrio* bacteria in the plastisphere, which degrade plastics to supply electron donors [17]. Despite growing awareness, there is an urgent need to focus on the presence, abundance, fluxes, and impacts of MNPs on element cycling and terrestrial biota [18]. MNPs accumulate across ecosystems, with evidence of ingestion by various organisms and accumulation in plants, making them accessible to organisms across trophic levels [14]. However, the **trophic transfer** of MNPs in terrestrial ecosystems, from soil to primary producers to various consumers, is complex. This complexity arises from the intricate dynamics of cellular-level uptake and **bioaccumulation**, which are influenced by the properties of MNPs, including their type, size, and concentration [19]. Primary producer plants take up 200 nm MNPs through their roots and 100 nm nanoplastics through their leaves [20,21]. The accumulation of MNPs in plant systems can serve as a key vector for their movement into AG-BG food webs [22]. In addition, MNPs can penetrate plant tissues, causing intercellular damage [19], ageing, BG herbivore damage, and mechanical injury [23]. Increasing MNP pollution in terrestrial ecosystems can further alter plant–herbivore interactions [24]. The entry of MNPs into plant systems may act as a major gateway for their transfer into the plant-associated AG-BG biota (Figure S3 in the supplemental information online), causing oxidative stress, tissue damage, behavioural changes, reproductive alterations, and metabolic disturbances [25,26].

To better understand the ecological impact of MNPs on AG-BG food webs and ecosystem multifunctionality, we made a comprehensive overview of MNPs ingestion, bioaccumulation, and their **ecotoxicological endpoints** on plants and the associated key AG-BG biota across trophic levels (Figures S4 and S5 in the supplemental information online), (Figures 1–2 and Box 1). Our conceptual framework outlines the MNPs transfer pathways within the AG-BG food webs (Figure 3, Key figure), highlighting their potential to disrupt biotic interactions and impact terrestrial ecosystem biodiversity and function (Figure 4).

Bioaccumulation and ecotoxicity of MNPs

Key drivers and patterns of bioaccumulation of MNPs in terrestrial biota

The key factors that drive the bioaccumulation of MNPs by plants include their size, type, concentration, and charge. For example, polystyrene (PS) MNPs (100–700 nm, 10 mg/ml) can enter cucumber roots and move to the aerial parts [27]. In wheat, PS and polyvinyl chloride (PVC) MNPs (30 nm, 10 mg/l) accumulate primarily in the root tips and surfaces, with PVC showing a high prevalence [28]. Both positively and negatively charged MNPs accumulate in the root tips of *Arabidopsis thaliana*; however, relatively fewer positively charged MNPs accumulate there [29]. These findings suggest that the bioaccumulation of MNPs in plants is strongly influenced by the identity of the plant and the characteristics of the MNPs [21,30]. The bioaccumulation of airborne MNPs in plant (AG) shoots can occur through stomata, enabling their entry into plant systems [31]. MNPs enter plants via transpiration (Figure S4 and Table S1 in the supplemental information online) and are translocated from the AG (leaf) to the BG (root) via vascular bundles [32]. MNPs can enter through the root cell wall (Figure S5) and can be transported via vascular assemblies [33–35], while also aggregating, penetrating, and accumulating on the root hairs and tips of plants, such as arabidopsis, wheat, and lettuce [35,36]. Key factors that drive the ingestion and bioaccumulation of MNPs by animals include (i) the availability of MNPs, (ii) the similarity of prey and MNPs (size, colour, or shape), (iii) feeding habits (generalist versus specialist), and (iv) nutritional state (starvation level) [37–39]. Most studies on the bioaccumulation of MNPs in AG biota have focused on the soft tissues of birds and insects (Table S2 and Figure S4 in the supplemental information online), with the gastrointestinal tract being the primary route for accumulation in vertebrates [40]. In the BG biota, most MNP bioaccumulation studies have focused on earthworms and nematodes (Table S3 in the supplemental information online). MNPs accumulate in the whole body of BG biota, as well as in the cells of yeasts and filamentous fungi [41,42] (Figure S5). Large MNPs (too

Glossary

Bioaccumulation: the net uptake of a contaminant (such as MNPs or additives) from the environment in all possible ways (contact, ingestion, and respiration) from abiotic and biotic sources (such as water, soil, and prey). Bioaccumulation is the most frequently used concept in ecological risk assessments for determining the range of pollutant dissemination within food webs.

Biomagnification: bioaccumulation in the primary producer (or prey) and the subsequent trophic transfer of a contaminant may lead to biomagnification at high trophic levels. Biomagnification means a high concentration of a contaminant (such as MNPs or additives) in consumers or predators than in their prey.

Cytotoxicity: the degree to which a toxic substance can damage a cell.

Ecological multifunctionality: the ability of an ecosystem to provide multiple functions and services. Many methods have been developed to evaluate ecological multifunctionality, with the most widely used being the averaging and threshold approaches.

Ecotoxicological endpoints: the values derived from ecological and toxicity tests, which are the results of specific measurements of the state or dynamics of an organism or other levels of biological organisation made during or after a contamination test. Ecotoxicological endpoints often include mortality, toxicity, physiology, reproduction, behaviour, and community properties.

Genotoxicity: the degree to which a toxic substance damages genetic information within a cell.

Micro/nanoplastics (MNPs): in 2004, the marine ecologist Richard Thompson introduced the concept of microplastics (MPs). In 2008, the National Oceanic and Atmospheric Administration proposed the most frequently used definition of MPs as plastic particles <5 mm in diameter. Nanoplastics are considered an extension of MPs. They have a high specific surface area and volume ratio and are highly reactive and toxic, with sizes ranging from 1 to <1000 nm. In this review, we use MNPs to represent plastics with diameters <5 mm.

Plastic additives: chemical substances added to plastics to improve their properties. These include nine functional classes of specialty plastic additives: plasticisers, antioxidants, antistatic agents, chemical blowing agents, flame large to pass through epithelial cells) appear to be excreted directly after ingestion. In general, smaller MNPs have a high bioaccumulation potential in AG-BG biota.

Ecotoxicological effects of MNPs on AG-BG biota

MNPs exposure causes toxicity and affects the population growth and diversity of AG biota (Figure 1). The intrinsic properties of MNPs, plant species, and environmental factors determine

retardants, heat stabilisers, impact modifiers, light stabilisers, and lubricants/slip additives. MNPs are toxic to many organisms in the natural environment when plastic additives are used.
Trophic transfer: the process by which elements, including contaminants, move from a lower to a higher trophic level.

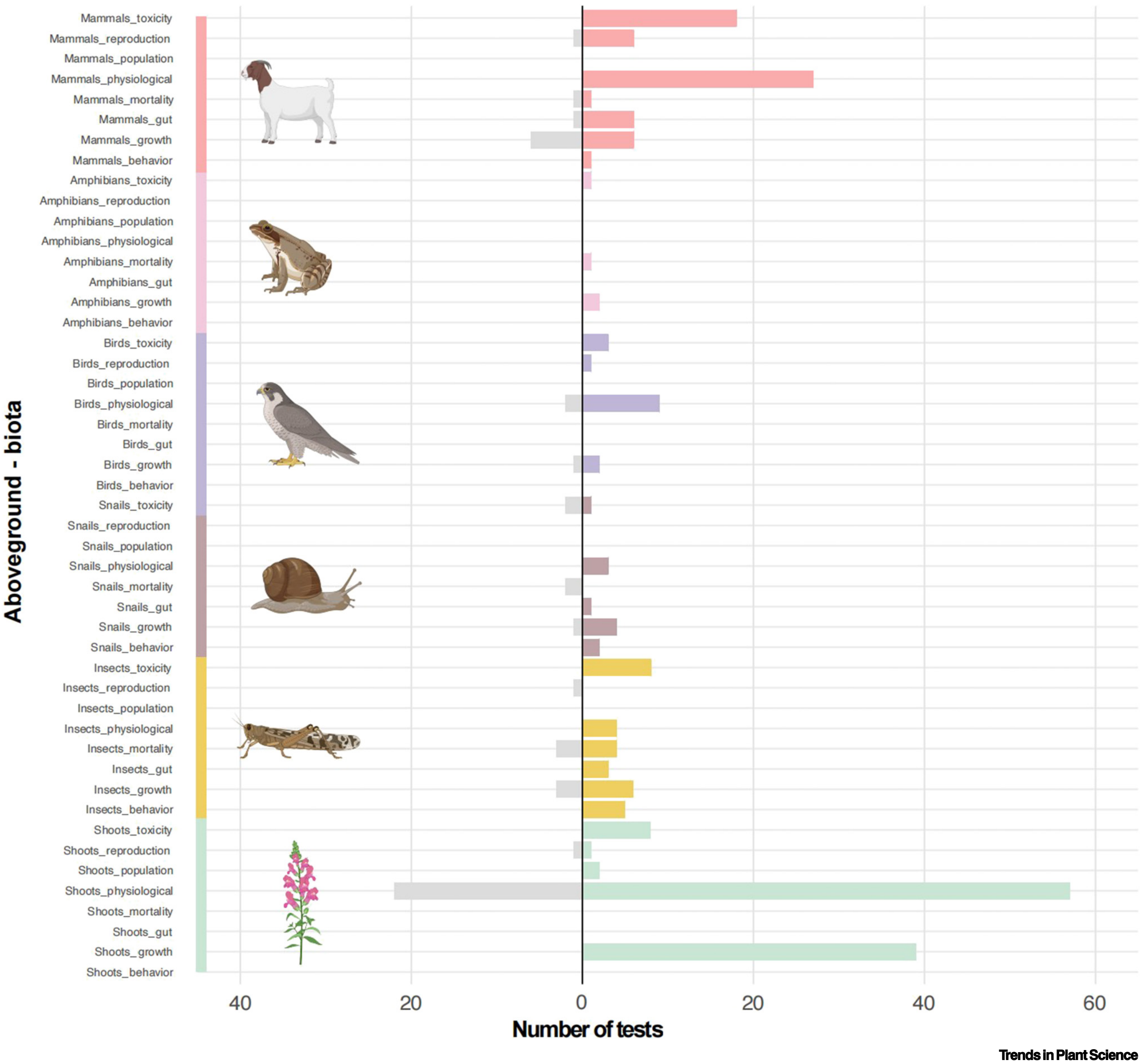
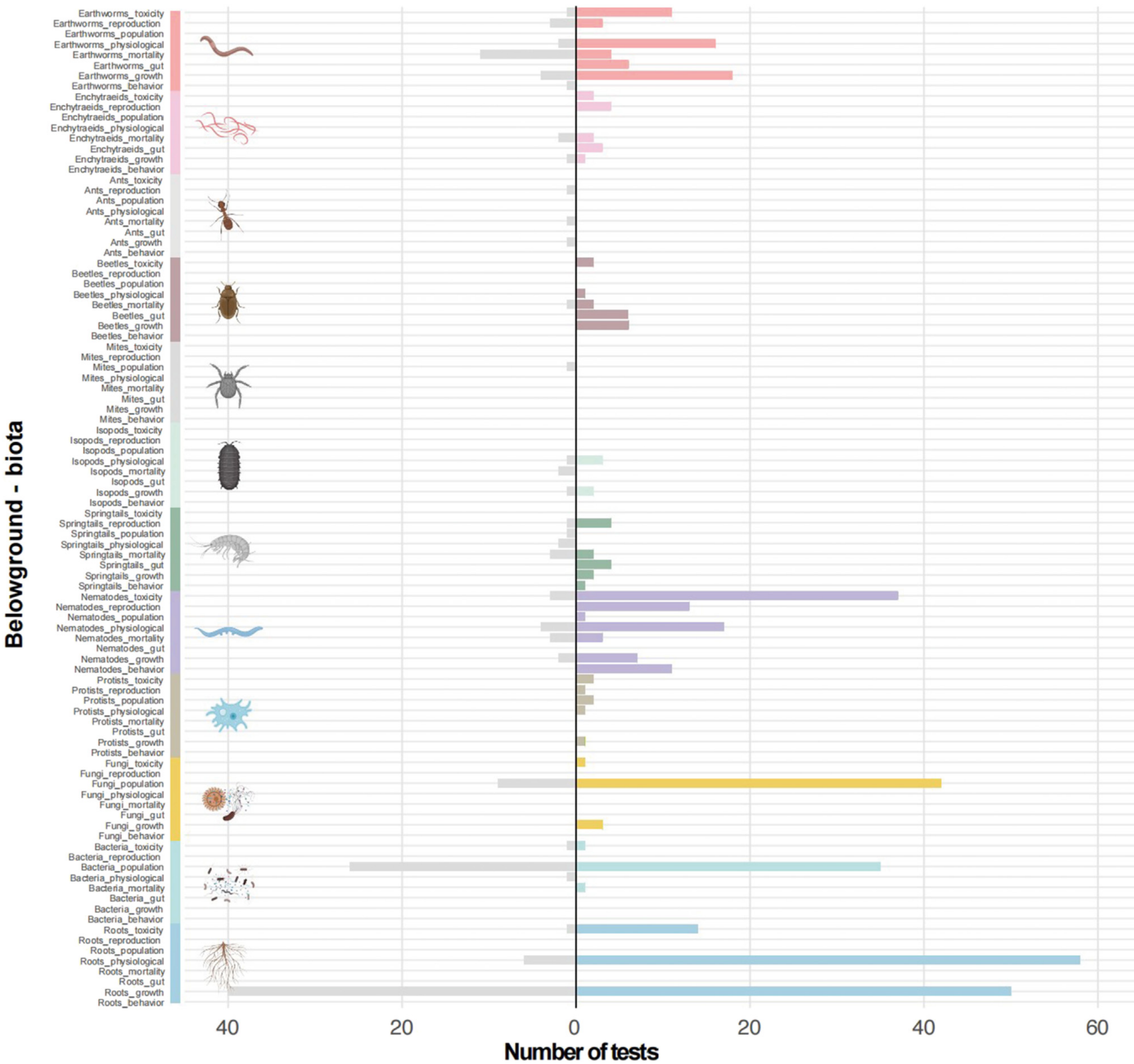


Figure 1. Ecotoxicological endpoints report of the effects of micro/nanoplastics (MNPs) on terrestrial aboveground (AG) biota. Each bar shows the total number of tests for the studied biota. The different colours of the bars, along with the pictures, represent the different groups of biota. The grey bar (left of the central line) represents the number of tests reporting nonsignificant effects of MNPs on specific ecotoxicological endpoints, whereas the coloured bars (right of the central line) represent the number of tests reporting significant effects of MNPs on specific ecotoxicological endpoints. For example, the top line represents 18 tests on the toxicity of MNPs in mammals (Mammals_toxicity), and all showed significant effects (the red bar to the right of the central line).



Trends in Plant Science

Figure 2. Ecotoxicological endpoints report of the effects of micro/nanoplastics (MNPs) on terrestrial belowground (BG) biota. Each bar shows the total number of tests for the studied biota. The different colours of the bars, along with the pictures, represent the different groups of biota. The grey bar (left of the central line) represents the number of tests reporting nonsignificant effects of MNPs on specific ecotoxicological endpoints, whereas the coloured bars (right of the central line) represent the number of tests reporting significant effects of MNPs on specific ecotoxicological endpoints. For example, the top line represents 12 tests on the toxicity of MNPs on earthworms (Earthworm_toxicity), with one nonsignificant result (the grey bar, left) and 11 significant effects (the coloured bar, right).

their toxic effects on plants [43]. Most studies have shown that MNPs negatively influence plant health by reducing the seed germination rate, inhibiting plant growth, decreasing chlorophyll content, and upregulating stress indicators at environmentally realistic concentrations [44], whereas a few studies have shown no or positive effects of MNPs on plants [43]. Different mechanisms are involved in the impact of MNPs on plant performance, including indirect effects, such as altering soil microbial structure and metabolism, and direct effects, such as serving as root barriers or

Box 1. Specific findings of bioaccumulation and ecotoxicity in terrestrial biota

Bioaccumulation of MNPs in plants

To date, 20 studies comprising 42 tests on 12 plant species (eight dicots and four monocots) have investigated the uptake and bioaccumulation of MNPs. The majority of the studies (37 out of 42 tests) demonstrated the accumulation and uptake of MNPs in the shoots and roots (Table S1 in the supplemental information online).

Bioaccumulation of MNPs in AG biota

For AG biota, 32 studies examined the ingestion or bioaccumulation of MNPs, encompassing 74 tests on 67 species. Birds were the most frequently studied group (35 tests), followed by insects (17 tests), mammals (ten tests), snails (five tests), reptiles (three tests), amphibians (three tests), and slugs (one test). Overall, MNPs were ingested (12 tests) or bioaccumulated (62 tests) in the AG biota (Table S2 in the supplemental information online).

Bioaccumulation of MNPs in BG biota

For BG biota, 35 studies investigated the ingestion and bioaccumulation of MNPs, involving 42 tests on 22 species. Earthworms (seven species) were the most studied group, followed by nematodes (one species), enchytraeids/worms (one species), fungi (three species), ants (two species), isopods (one species), soil protists (two species), and springtails, beetles, centipedes, scorpions, and spiders (one species each). Most tests (37 out of 42) showed that MNPs could be ingested (11 tests) or bioaccumulated (26 tests) in the BG biota (Table S3 in the supplemental information online).

Ecotoxicological effects of MNPs on AG biota

The ecotoxicological studies of MNPs on AG biota were mostly short-term (82%) and laboratory-scale (98%) (Figure S6A,B in the supplemental information online). Ecotoxicological endpoints followed the categorisation methods of Green *et al.* [97], which define 'toxicity' as the testing of sublethal responses, such as cytotoxicity, immunotoxicity, genotoxicity, and mutagenicity. We have included gut microbiota as a new endpoint. A total of 101 papers identified 511 ecotoxicological endpoints on 59 AG species (Figure S6C–E). MNPs significantly affected the physiology of the majority AG biota (407 out of 511 tests) (Figure S7A in the supplemental information online). Growth/malformation was the most frequently tested endpoint, followed by physiology, toxicity, mortality, reproduction, gut microbiota, behaviour, and population/diversity (Figure S7B).

Ecotoxicological effects of MNPs on BG biota

Ecotoxicological studies on the BG biota were also short-term (84%) and laboratory-based (76%) (Figure S8A,B in the supplemental information online). A total of 162 papers identified 659 ecotoxicological endpoints on 83 BG species (Figure S8C–E). MNPs caused significant changes (522 out of 659 tests) in the BG biota, particularly in growth, population/diversity, and physiology (Figure S9 in the supplemental information online).

accumulating within plants [45]. In insects, MNPs cause toxicity and affect behaviour, physiology, and gut microbiota (Figure 1), whereas one study has shown no effect on reproduction, growth, or mortality [46]. MNPs can induce biomolecular and biochemical responses in insects [47] or indirectly alter the gut microbiota [48]. MNPs can lower snail movement and gut microbe diversity but were not found to impact mortality [49,50]. In birds, MNPs are toxic and affect reproduction and physiology. MNPs can also reduce growth and cause malformations in amphibians (Figure 1) [51]. The bioaccumulation of MNPs in various mammalian tissues induces toxicity and affects physiology, behaviour, gut microbiota, and reproduction (Figure 1; Figures S6 and S7 in the supplemental information online) [33], while the effect on growth is still debated.

BG, MNPs show complex ecotoxicological effects on plant roots, soil microorganisms, and soil animals (Figure 2). Exposure to MNPs induces **cytotoxicity** and **genotoxicity** in meristematic tissues, and affects root physiology and development. Interestingly, half of the studies demonstrated positive effects of MNPs on root length, whereas the other half showed negative effects [44]. These effects can be positive or negative depending on the concentration, size, and type of MNPs [33,34]. For example, the root length of monocots increased when exposed to 1 μm PS and polyethylene MNPs at 0.1 mg/l but decreased with exposure to 0.02 μm MNPs at 50 mg/l [44]. The root biomass of

Key figure

Schematic representation of the transport of micro/nanoplastics (MNPs) in aboveground (AG) and belowground (BG) food webs

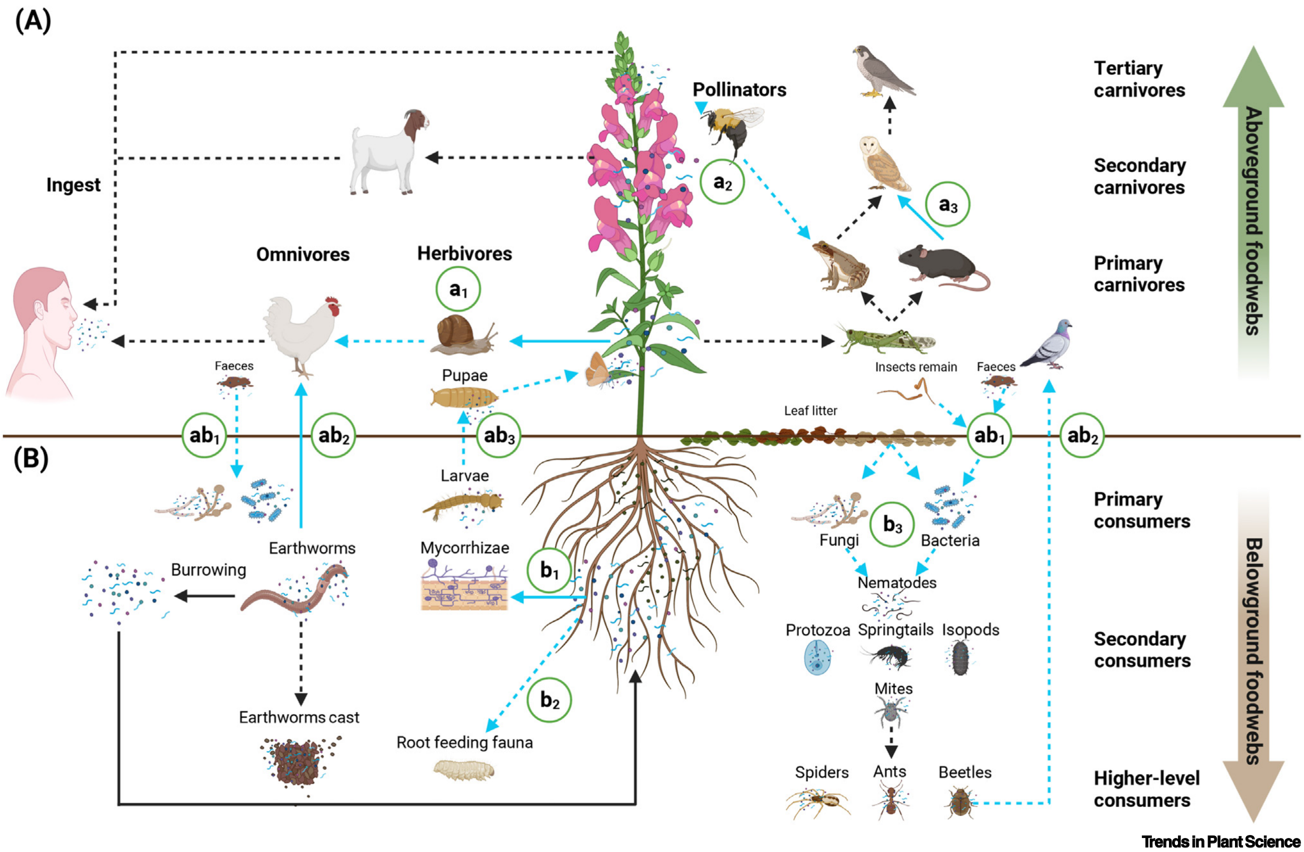


Figure 3. (A) Transfer of MNPs along the AG food web. The accumulation of MNPs in AG food webs is transferred from lower trophic levels (producers) to higher levels (top predators) through herbivory, uptake, and accumulation, which may interfere with plant–insect interactions, herbivore–carnivore interactions, pollinators, and natural enemies [35,49,66]. MNPs can be transferred along AG food webs via the following three main pathways: (a1) leaf–herbivores, (a2) flower–pollinators, and (a3) prey–predators. (B) Transfer of MNPs along BG food webs. Feeding and accumulation of MNPs in soil biota promote the transport of MNPs between BG food webs, inhibiting nutrient turnover, root performance, and energy flow, thus indirectly affecting AG food webs [57,96]. MNPs can be transferred along BG food webs via the following three main pathways: (b1) root–arbuscular mycorrhizal fungi, (b2) root–herbivores, and (b3) leaf litter–decomposers. MNPs can be transferred between AG and BG food webs via the following three main pathways: (ab1) AG litter/remains–BG decomposers, (ab2) BG prey–AG predators, and (ab3) BG larvae–AG adults. The light blue unbroken (already reported) and broken (not reported) arrows represent the nine key pathways of MNPs and different nodes in the trophic webs. Black unbroken and broken arrows indicate other possible transfers of MNPs along AG and BG food webs. The schematic diagram was created using BioRender.

Lolium perenne was significantly increased by high-density polyethylene exposure but reduced by polylactic acid exposure [52]. MNPs caused significant physiological changes in most plant roots; however, their effects on root growth were inconsistent in laboratory experiments. Given the limited evidence on the impact of environmentally realistic concentrations of MNPs, standardised testing at environmentally relevant levels is essential to understand their ecological risks. MNPs have been found to alter the microbial community composition and diversity in most studies (Figure 2). For example, MNPs can increase the total microbial biomass and change soil microbial activity [53]. MNPs were shown to increase the number of Gram-negative bacteria but decrease the number of Gram-positive bacteria by decreasing oxygen concentrations [54]. MNPs increase the

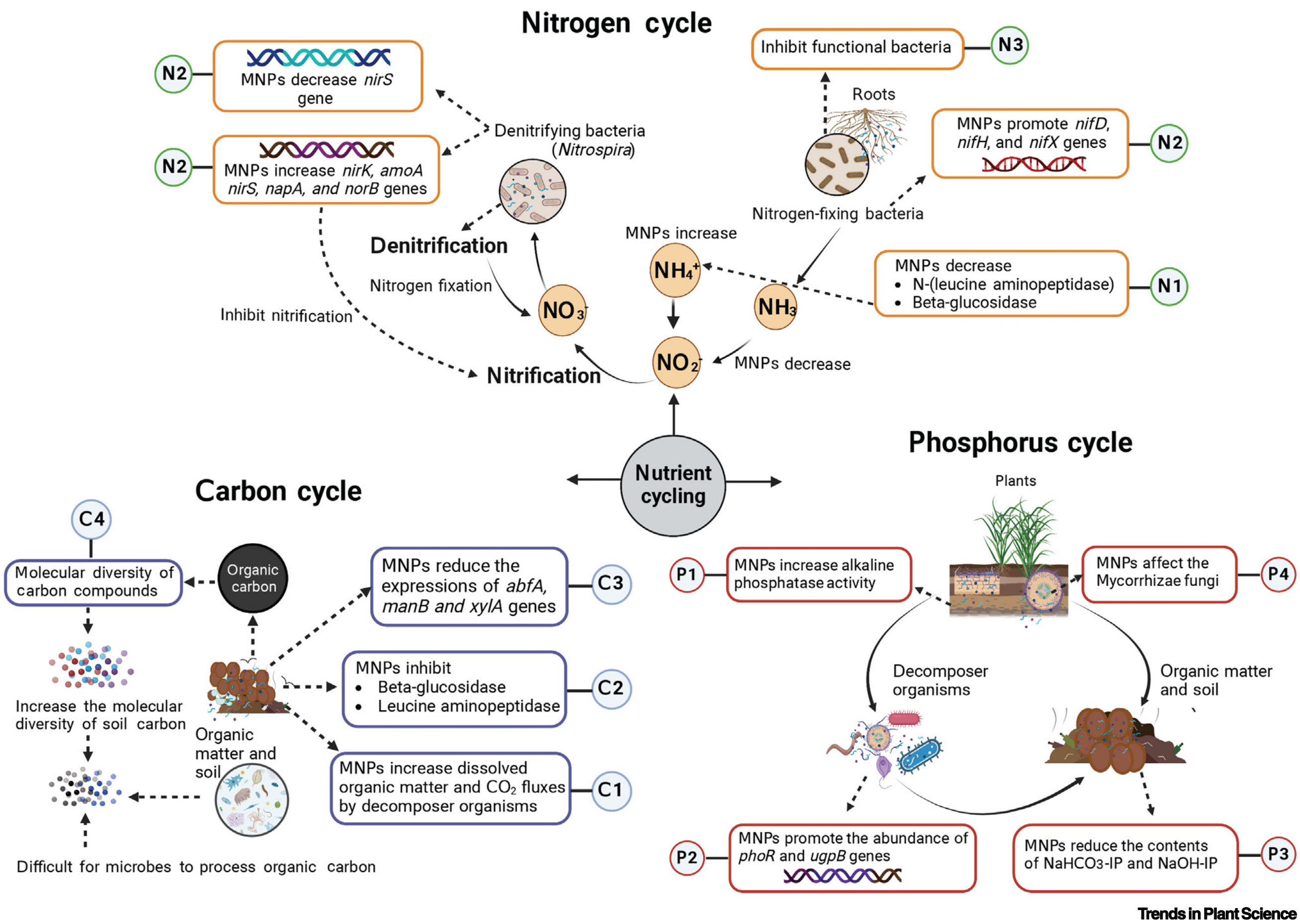


Figure 4. Ecological effects of micro/nanoplastics (MNPs) on carbon, nitrogen, and phosphorus cycling. The unbroken arrow indicates the natural process of nutrient cycling, whereas the broken arrow represents the reported effects of MNPs on nutrient cycling. The main mechanisms for the effects of MNPs on the carbon cycle include (C1) increased dissolved organic matter and CO_2 fluxes by microorganisms [82], (C2) inhibition of enzymatic activities [81], (C3) reduction of functional genes [80], and (C4) functional complexity of soil organic carbon [84]. The main mechanisms for the effects of MNPs on the nitrogen cycle include: (N1) reduction in enzymatic activities [85], (N2) inhibition of different functional genes [86,87], and (N3) changes in microbial activity, abundance, and structure [85]. The main mechanisms for the effects of MNPs on the phosphorus cycle include: (P1) promotion of enzymatic activities [86], (P2) increased functional gene abundance [86], (P3) reduction in the contents of $NaHCO_3$-IP (inorganic phosphorus) and NaOH-IP through phosphorus-related microorganisms [86], and (P4) indirect effects on the phosphorus cycle through mycorrhizal fungi [93,94]. The schematic diagram was created using BioRender.

abundance of fungal hyphae [53] but decrease diversity at high concentrations [55]. Soil MNPs may also serve as a habitat for microorganisms [56], attracting specific microbial taxa (such as Nocardioidaceae) while inhibiting some sensitive microbes (including Nitrospirales) [57].

MNPs can affect the survival, growth, reproduction, and gut microbiota of soil fauna. Earthworms and nematodes are among the most frequently observed soil fauna (Figure 2; Figures S8 and S9 in the supplemental information online). In earthworms, MNPs cause toxicity and affect gut microbiota and physiology, but no significant effects on behaviour have been observed. MNPs significantly affect nematode reproduction, behaviour, and physiology and cause toxicity (Figure 2). MNPs can also modify the population dynamics of nematodes [58] by reducing their growth and reproductive rates [59]. In springtails, MNPs exposure can lead to behavioural changes [42], enhance gut bacterial diversity, and reduce growth and reproduction [43,44], ultimately reducing populations and increasing mortality [60].

MNPs can affect the growth and gut microbiota of beetles and cause mortality (Figure 2). The effects on the feeding activity of soil woodlice depend on the shape and type of MNPs [61]. However, fewer studies are available on soil predators, such as spiders, mites, and ants.

In summary, MNPs bioaccumulate across plants and the associated AG-BG biota, causing ecotoxicological effects at multiple trophic levels (Figures 1, 2, S4, and S5). Most studies have been short-term laboratory experiments (Figure S6A, B), highlighting the need for long-term field research to understand the true impact of MNPs on natural ecosystems. There is also a significant gap in our understanding of how MNPs affect AG-BG communities and trophic interactions.

Trophic transfer of MNPs

Our understanding of MNPs trophic transfer and **biomagnification** in AG-BG food webs is limited because of the lack of field experiments. Aquatic biota may ingest MNPs unintentionally and may be highly affected by their partitioning, whereas the uptake of MNPs by terrestrial animals may depend more on predator–prey interactions, making the biomagnification of MNPs through trophic transfer along terrestrial food webs more likely [6]. We propose key trophic and non-trophic MNPs transfer pathways within and between the AG-BG food webs (Figure 3) detailed in pathways a1–a3, b1–b3, and ab1–ab3.

We emphasise three key AG pathways for the trophic transfer of MNPs along AG food webs:

(i) (a1) Leaves–herbivores. The airborne deposition of MNPs on plant leaves can negatively affect the feeding and foraging speeds of African giant snails (*Achatina fulica*) by transferring MNPs through chewing and ingestion [49]. Additionally, MNPs have been transferred from *Lactuca sativa* (lettuce) leaves (0.73 to 15.6 μg/g) to *Bradybaena ravida* snails (0.33 to 10.7 μg/kg) through feeding, with a translocation factor of 0.45, indicating trophic dilution within the AG food web [62]. MNPs have also been shown to be transferred from plants to insects in freshwater environments [63] (Figure 3A).
(ii) (a2) Flowers–pollinators. MNPs can adhere to the surface of flowers [27] and may be ingested or accumulated in the midguts of pollinators (including bees and butterflies) [64].
(iii) (a3) Prey–predators. MNPs can also be transferred via prey–predator pathways (Figure 3A). For instance, MNPs found in the digestive tract of red-shouldered hawks (*Buteo lineatus*) likely originated from prey, such as small mammals and snakes [65]. The highest MNPs levels have been found in the predatory barn owl (*Tyto alba*), whereas the lowest have been found in the herbivorous Savi pine vole (*Microtus savii*) [66]. Similarly, MNPs accumulated in predatory spiders (*Pardosa pseudoannulata*) after ingesting MNPs-contaminated prey mosquitoes (*Culex quinquefasciatus*) [67].

The three untested hypothetical BG pathways are:

(i) (b1) Roots–arbuscular mycorrhiza fungi. MNPs may affect the symbiotic association between plant roots and mycorrhizal fungi [68], potentially facilitating MNPs dispersal across BG food webs (Figure 3B). MNPs have been proven to accumulate in yeasts and filamentous fungi [23]. Similarly, in axenic growth systems, arbuscular mycorrhizal hyphae capture MNPs and transfer them to plant roots [69], suggesting that the transfer of MNPs between roots and arbuscular mycorrhizal fungi is possible.
(ii) (b2) Roots–herbivores. Certain root herbivores, such as aphids [70], can accumulate MNPs [71,72], indicating that MNPs can be transferred from roots to root herbivores through feeding activities (Figure 3B).

(iii) (b3) Leaf litter–decomposers. Decomposers can ingest and accumulate MNPs while decomposing leaf litter and organic matter [73]. Soil omnivores and predators may absorb and accumulate MNPs by feeding on soil fungi and bacteria [74]. Furthermore, MNPs ingested by soil omnivores or predators might be transferred back to soil decomposers through their faecal pellets, potentially disrupting energy flow in BG food webs (Figure 3B).

Given the interdependence of AG and BG food webs [75], MNPs could be transferred between them via three hypothetical AG–BG pathways:

(i) (ab1) AG litter/remains–BG decomposers. MNPs in AG litter, such as bird faeces and insect remains, may be transferred to the BG food web through feeding and degradation by soil biota (Figure 3).
(ii) (ab2) BG prey–AG predators. Earthworms, as BG prey, can ingest and accumulate MNPs, transferring them to AG predators, such as chickens, suggesting the potential for MNPs transfer from BG biota to AG biota [76]. This transfer pathway may also affect humans through food consumption.
(iii) (ab3) BG larvae–AG adults. MNPs may be transferred from BG larval insects to AG food webs during metamorphosis [77]. For example, mosquitoes transfer MNPs ontogenetically from larval to adult life stages [78]. Similarly, *Tenebrio molitor* larvae (yellow mealworms) transfer MNPs to mice via feeding, highlighting the possibility for trophic transfer from BG larvae to AG adults [79]. Given the widespread presence of larvae of insects, such as Coleoptera and Diptera, in soils, we propose that transporting MNPs from the BG to AG food webs through the larval–adult pathway is significant.

In summary, MNPs can travel through various pathways in terrestrial ecosystems, with plants being the key entry points. Once absorbed by plants, MNPs interact with herbivores, pollinators, and mycorrhiza. However, many of these transfer pathways remain hypothetical and require empirical validation, and the long-term impact of MNPs on ecosystem functions remains largely unknown.

Impacts of MNPs on terrestrial ecosystem multifunctionality

MNPs can change soil properties (Figure S10 in the supplemental information online), posing significant challenges to ecological multifunctionality and services [18]. We identified how MNPs affect the cycling of nutrients, including carbon, nitrogen, and phosphorus, through enzymatic activities, functional genes, and soil microorganisms (Figure 4), with effects varying according to the type, size, concentration, and soil characteristics of MNPs. MNPs are carbon-based materials that should be treated as emergent components of carbon pools because the quantities of plastic stock and fluxes in certain ecosystems rival those of natural organic carbon [3,80]. By altering the microbial community, MNPs increase the dissolved organic matter and CO_2 fluxes (Figure 4, C1). MNPs have been reported to inhibit the enzymatic activities of β-glucosidase and leucine aminopeptidase [81], suggesting potential negative impacts on carbon cycling (Figure 4, C2). MNPs can also downregulate genes involved in hemicellulose (*abfA*, *manB*, and *xylA*) and starch (*sga*) degradation (Figure 4, C3), potentially elevating the microbial metabolic quotient and thereby enhancing CO_2 emissions from soils [82,83]. A global meta-analysis indicated that MNPs increase the soil carbon pool and microbial biomass, thereby promoting soil CO_2 emissions [15]. However, CO_2 emissions typically depend on MNPs concentrations. For instance, a high concentration of low-density polyethylene (1.00%) stimulates CO_2 release, whereas low concentrations (0.01% and 0.10%) have no significant effect on CO_2 emissions in agricultural soil [82]. In addition, MNP-absorbing chemicals can increase the molecular diversity of soil carbon (Figure 4, C4) and its resistance to microbial degradation [84].

MNPs may affect the assimilation process of nitrogen by decreasing extracellular enzyme activity (Figure 4, N1) [85]. Polylactic acid MNPs promote the abundance of genes associated with nitrogen fixation (*nifD, nifH, and nifX*) and denitrification (*nirS, napA, and norB*), while suppressing nitrification pathways. This shift results in the significant accumulation and release of ammonia nitrogen in the soil (Figure 4, N2) [86]. During the denitrification stage, low-density polyethylene-MNPs promote the abundance of genes related to denitrifying bacteria [86,87]. For example, low concentrations of polyamide MNPs (0.3%) increase the abundance of *nirS* and *nirK* genes, whereas high concentrations (1%) decrease *nirK* gene abundance, suggesting that low concentrations of polyamide MNPs enhance denitrification, whereas high concentrations may inhibit this process [88]. MNPs can also influence the composition, community structure, and number of microorganisms, further affecting nutrient cycling (Figure 4, N3) [85]. For instance, MNPs enhance the activities of soil catalase and urease, which in turn reshape the soil bacterial community composition [89]. These shifts prominently affect the microbial composition involved in nitrogen cycling, including key bacterial taxa, such as *Burkholderiaceae*, *Xanthobacteraceae*, and *Pseudomonadaceae* [90,91]. MNP-induced changes in soil properties, such as pH, organic matter, redox potential, and bulk density, can also drive shifts in the microbial community structure, altering nitrogen cycling processes, such as denitrification, nitrate reduction, and organic matter decomposition [91]. Furthermore, MNPs significantly affect the structure of the soil microbial network, potentially modifying microorganism interactions and influencing the rate and efficiency of nitrogen cycling [91,92].

In addition, MNPs increase alkaline phosphatase activity (Figure 4, P1), promote the expression of phosphorus regulation (*phoR*) and organophosphorus mineralisation (*phoD*) genes (Figure 4, P2), stimulate the release of sedimentary phosphorus, and influence the phosphorus cycle [86]. MNPs can reduce NaHCO -IP (inorganic phosphorus) and NaOH-IP by changing phosphorus-related microbial communities (Figure 4, P3) [86]. For instance, PVC-MNPs significantly increase the relative phosphorus-solubilising microbes in acid-loamy soils, resulting in greater acid phosphatase activity in MNP-amended soils [90]. MNPs affect arbuscular mycorrhizal fungi (Figure 4, P4) [93], thereby influencing soil phosphorus [94]. At low concentrations, MNPs increase soil fungi and enzyme activity involved in soil phosphorus cycling, whereas at high concentrations, they reduce available phosphorus and total phosphorus in the soil, disrupting phosphorus cycling [95]. Further research is required to fully understand the effects of MNPs on global nutrient cycling. It is important to note that previous research has focused only on the influence of MNPs on one or a few ecosystem functions. There remains a large research gap in comprehensively evaluating the effects of MNPs on ecosystem multifunctionality. Furthermore, few studies have examined the effects of MNPs on the relation between biodiversity and ecological multifunctionality.

Concluding remarks

Emerging research suggests that MNPs, similar to those observed in marine food webs, may be widely present and have a significant impact on terrestrial biota. MNPs can accumulate in various terrestrial plants and their associated AG-BG biota, travel through multiple pathways within terrestrial ecosystems, and potentially affect biodiversity and ecosystem multifunctionality. We highlight the critical research gaps in understanding the long-term effects, trophic transfer, and overall impacts on ecosystem multifunctionality (see Outstanding questions). Urgent and comprehensive studies on terrestrial ecosystems are needed to fully assess the ecological impacts of MNPs and develop strategies for mitigating their effects on plants and their interconnected food webs. While further evidence is required, existing data support the adoption of a precautionary approach to regulate MNPs pollution in terrestrial environments.

Outstanding questions

Research on MNPs in terrestrial ecosystems remains in the early stages of development. However, the available data on their ecological effects on plants and the associated AG-BG food webs suggest a range of potential impacts on terrestrial ecosystems. Answering the following questions would be of interest to the scientific community, policymakers, and the general public:

Effects of MNPs on plants and AG-BG biota. How do MNPs influence plant root architecture, nutrient uptake, and overall plant fitness and productivity? Can MNPs be translocated throughout the plant? What are the potential implications for plant health and food chain safety? What are the evolutionary consequences of prolonged MNP exposure within plants and the associated AG-BG biota?

Effects of MNPs on interactions within AG-BG food webs. How do MNPs affect plant–microbe interactions within the rhizosphere? What are the impacts of MNPs on plant–pollinator dynamics and the overall reproductive success of plants? How do MNPs affect AG-BG biota interactions, including mutualism, commensalism, competition, and predation, across various trophic levels?

What are the best approaches for testing the proposed nine trophic and non-trophic transfer pathways of MNPs within and between AG-BG food webs? What are the mechanisms and rates of MNP trophic transfer and biomagnification across interconnected AG and BG food webs? Can MNPs with **plastic additives** (beyond this review's scope) bioaccumulate or biomagnify from plants to higher trophic levels in AG–BG food webs?

Effects of MNPs on biodiversity-ecosystem multifunctionality. What are the long-term effects of MNPs on plant community composition? How do MNPs influence various aspects of biodiversity, including species, functional, and genetic diversity in plants and the AG-BG biota? What are the potential future impacts of MNPs on ecosystem multifunctionality in terrestrial ecosystems, such as nutrient cycling and litter decomposition? How do environmentally relevant concentrations of MNPs affect the relation between biodiversity and ecosystem multifunctionality in microcosm experiments and field manipulations?

Data availability

The datasets analysed in the current study, including the specific information on tested species, MNPs characteristics, and ecotoxicological endpoints (101 papers identified 511 ecotoxicological endpoints on 59 AG species, and 148 papers identified 659 ecotoxicological endpoints on 96 BG species) are available from ScienceDB.

Acknowledgments

This work is supported by the National Key R&D Program of China (2022YFF1302401), the National Natural Science Foundation of China (32371716), the Youth Innovation Promotion Association of the Chinese Academy of Sciences (2022401), the Yunnan Revitalization Talents Support Program–Young Talent Project (XDYC-QNRC-2022-0025), Yunling Scholars, Yunnan Fundamental Research Projects (grant NO. 202201AT070700), the Key R&D Program of Yunnan Province (202403AC100028), the NSFC China–US Dimensions of Biodiversity Grant (DEB: 32061123003), Western Light Foundation of Regional Development Fund and the 14th Five-Year Plan of the Xishuangbanna Tropical Botanical Garden, Chinese Academy of Sciences (XTBG-1450101 and XTBG-1450102). E.G.X. thanks the support of the Department of Biology, University of Southern Denmark, and Danmarks Frie Forskningsfond (0165-00056B). Muhammad Ilyas thanks the support of the China Scholarship Council (CSC). We thank Yunyun He, who helped us to improve the figures. We also thank Professor Thijs Bosker, whose comments improved this manuscript.

Declaration of interests

No interests are declared.

Supplemental information

Supplemental information associated with this article can be found online at https://doi.org/10.1016/j.tplants.2024.11.018.

References

1. Smith, J. and Vignieri, S. (2021) A devil's bargain. *Science* 373, 34–35
2. Simon, N. *et al.* (2021) A binding global agreement to address the life cycle of plastics. *Science* 373, 43–47
3. Stubbins, A. *et al.* (2021) Plastics in the Earth system. *Science* 373, 51–55
4. Geyer, R. *et al.* (2017) Production, use, and fate of all plastics ever made. *Sci. Adv.* 3, 1700782
5. Thompson, R. *et al.* (2005) New directions in plastic debris. *Science* 310, 1117
6. Huang, D. *et al.* (2022) Recent advances on the transport of microplastics/nanoplastics in abiotic and biotic compartments. *J. Hazard. Mater.* 438, 129515
7. Law, K.L. and Thompson, R.C. (2014) Microplastics in the seas. *Science* 345, 144–145
8. MacLeod, M. *et al.* (2021) The global threat from plastic pollution. *Science* 373, 61–65
9. Aves, A.R. *et al.* (2022) First evidence of microplastics in Antarctic snow. *Cryosph* 16, 2127–2145
10. Benson, N.U. *et al.* (2022) Micro(nano)plastics prevalence, food web interactions, and toxicity assessment in aquatic organisms: A Review. *Front. Mar. Sci.* 9, 851281
11. Büks, F. and Kaupenjohann, M. (2020) Global concentrations of microplastics in soils–a review. *Soil* 6, 649–662
12. Wei, H. *et al.* (2022) Meta-analysis reveals differential impacts of microplastics on soil biota. *Ecotoxicol. Environ. Saf.* 230, 113150
13. Nizzetto, L. *et al.* (2016) Pollution: do microplastics spill on to farm soils? *Nature* 537, 488
14. Thompson, R.C. *et al.* (2024) Twenty years of microplastics pollution research – what have we learned? *Science* 386, 6720
15. Zhao, S. *et al.* (2024) Microplastic pollution promotes soil respiration: a global-scale meta-analysis. *Glob. Chang. Biol.* 30, 17415
16. Huang, W. and Xia, X. (2024) Element cycling with micro (nano) plastics. *Science* 385, 933–935
17. Wang, H. *et al.* (2023) Stable isotopic and metagenomic analyses reveal microbial-mediated effects of microplastics on sulfur cycling in coastal sediments. *Environ. Sci. Technol.* 57, 1167–1176
18. Rillig, M.C. (2012) Microplastic in terrestrial ecosystems and the soil? *Environ. Sci. Technol.* 12, 6453–6454
19. Rose, P.K. *et al.* (2023) Microplastics and nanoplastics in the terrestrial food chain: uptake, translocation, trophic transfer, ecotoxicology, and human health risk. *TrAC Trends Anal. Chem.* 167, 117249
20. Xu, Z. *et al.* (2022) Toxic effects of microplastics in plants depend more by their surface functional groups than just accumulation contents. *Sci. Total Environ.* 833, 155097
21. Ilyas, M. *et al.* (2024) Foliar implications of polystyrene nanoplastics on leafy vegetables and its ecological consequences. *J. Hazard. Mater.* 480, 136346
22. Dong, X. *et al.* (2023) From natural environment to animal tissues: a review of microplastics(nanoplastics) translocation and hazards studies. *Sci. Total Environ.* 855, 158686
23. Ullah, R. *et al.* (2021) Microplastics interaction with terrestrial plants and their impacts on agriculture. *J. Environ. Qual.* 50, 1024–1041
24. Rondoni, G. *et al.* (2021) Microplastics alter behavioural responses of an insect herbivore to a plant–soil system. *Sci. Total Environ.* 787, 147716
25. Wang, J. *et al.* (2019) Polystyrene microplastics cause tissue damages, sex-specific reproductive disruption and transgenerational effects in marine medaka (*Oryzias melastigma*). *Environ. Pollut.* 254, 113024
26. Burgos-Aceves, M.A. *et al.* (2021) Physiological and metabolic approach of plastic additive effects: Immune cells responses. *J. Hazard. Mater.* 404, 124114
27. Li, Z. *et al.* (2021) The distribution and impact of polystyrene nanoplastics on cucumber plants. *Environ. Sci. Pollut. Res.* 28, 16042–16053
28. Zhu, J. *et al.* (2024) Accumulation of nanoplastics by wheat seedling roots: Both passive and energy-consuming processes. *J. Hazard. Mater.* 480, 136052
29. Sun, X.-D. *et al.* (2020) Differentially charged nanoplastics demonstrate distinct accumulation in *Arabidopsis thaliana*. *Nat. Nanotechnol.* 15, 755–760
30. Azeem, I. *et al.* (2021) Uptake and accumulation of nano/microplastics in plants: a critical review. *Nanomaterials* 11, 2935
31. Bi, M. *et al.* (2020) What roles are terrestrial plants playing in global microplastic cycling? *Environ. Sci. Technol.* 54, 5325–5327
32. Sun, H. *et al.* (2021) Foliar uptake and leaf-to-root translocation of nanoplastics with different coating charge in maize plants. *J. Hazard. Mater.* 416, 125854

33. Jiang, X. *et al.* (2019) Ecotoxicity and genotoxicity of polystyrene microplastics on higher plant *Vicia faba*. *Environ. Pollut.* 250, 831–838
34. Giorgetti, L. *et al.* (2020) Exploring the interaction between polystyrene nanoplastics and *Allium cepa* during germination: Internalization in root cells, induction of toxicity and oxidative stress. *Plant Physiol. Biochem.* 149, 170–177
35. Li, L. *et al.* (2020) Effective uptake of submicrometre plastics by crop plants via a crack-entry mode. *Nat. Sustain.* 3, 929–937
36. Taylor, S.E. *et al.* (2020) Polystyrene nano-and microplastic accumulation at *Arabidopsis* and wheat root cap cells, but no evidence for uptake into roots. *Environ. Sci. Nano* 7, 1942–1953
37. Santos, R.G. *et al.* (2021) Plastic ingestion as an evolutionary trap: toward a holistic understanding. *Science* 373, 56–60
38. Steen, R. *et al.* (2016) Plastic mistaken for prey by a colony-breeding Eleonora's falcon (*Falco eleonorae*) in the Mediterranean Sea, revealed by camera-trap. *Mar. Pollut. Bull.* 106, 200–201
39. Luo, Y. *et al.* (2023) Microplastics are transferred by soil fauna and regulate soil function as material carriers. *Sci. Total Environ.* 857, 159690
40. Schirinzi, G.F. *et al.* (2020) A new digestion approach for the extraction of microplastics from gastrointestinal tracts (GITs) of the common dolphinfish (*Coryphaena hippurus*) from the western Mediterranean Sea. *J. Hazard. Mater.* 397, 122794
41. Miyazaki, J. *et al.* (2015) Cytotoxicity and behavior of polystyrene latex nanoparticles to budding yeast. *Colloids Surf. A Physicochem. Eng. Asp.* 469, 287–293
42. Nomura, T. *et al.* (2016) Cytotoxicity and colloidal behavior of polystyrene latex nanoparticles toward filamentous fungi in isotonic solutions. *Chemosphere* 149, 84–90
43. Li, J. *et al.* (2022) Effects of microplastics on higher plants: a review. *Bull. Environ. Contam. Toxicol.* 109, 241–265
44. Zantis, L.J. *et al.* (2023) Nano-and microplastics commonly cause adverse impacts on plants at environmentally relevant levels: a systematic review. *Sci. Total Environ.* 867, 161211
45. Chen, Y. *et al.* (2024) Effects of microplastics on soil carbon pool and terrestrial plant performance. *Carbon Res.* 3, 37
46. Matthews, S. *et al.* (2021) Polystyrene micro-and nanoplastics affect locomotion and daily activity of Drosophila melanogaster. *Environ. Sci. Nano* 8, 110–121
47. Malafaia, G. *et al.* (2020) Polyethylene microplastics are ingested and induce biochemical changes in *Culex quinquefasciatus* (Diptera: Culicidae) freshwater insect larvae. *Ecotoxicol. Environ. Contam* 15, 79–89
48. Wang, K. *et al.* (2021) Gut microbiota protects honey bees (*Apis mellifera* L.) against polystyrene microplastics exposure risks. *J. Hazard. Mater.* 402, 123828
49. Chae, Y. and An, Y.-J. (2020) Nanoplastic ingestion induces behavioral disorders in terrestrial snails: trophic transfer effects via vascular plants. *Environ. Sci. Nano* 7, 975–983
50. Song, Y. *et al.* (2019) Uptake and adverse effects of polyethylene terephthalate microplastics fibers on terrestrial snails (*Achatina fulica*) after soil exposure. *Environ. Pollut.* 250, 447–455
51. Boyero, L. *et al.* (2020) Microplastics impair amphibian survival, body condition and function. *Chemosphere* 244, 125500
52. Boots, B. *et al.* (2019) Effects of microplastics in soil ecosystems: above and below ground. *Environ. Sci. Technol.* 53, 11496–11506
53. Fan, P. *et al.* (2022) Effects of different concentrations and types of microplastics on bacteria and fungi in alkaline soil. *Ecotoxicol. Environ. Saf.* 229, 113045
54. Zang, H. *et al.* (2020) Microplastics in the agroecosystem: are they an emerging threat to the plant–soil system? *Soil Biol. Biochem.* 148, 107926
55. Wang, F. *et al.* (2020) Interactions of microplastics and cadmium on plant growth and arbuscular mycorrhizal fungal communities in an agricultural soil. *Chemosphere* 254, 126791
56. Zhou, J. *et al.* (2021) The microplastisphere: Biodegradable microplastics addition alters soil microbial community structure and function. *Soil Biol. Biochem.* 156, 108211
57. Gao, B. *et al.* (2021) Microplastic addition alters the microbial community structure and stimulates soil carbon dioxide emissions in vegetable-growing soil. *Environ. Toxicol. Chem.* 40, 352–365
58. Mueller, M.-T. *et al.* (2020) Species-specific effects of long-term microplastic exposure on the population growth of nematodes, with a focus on microplastic ingestion. *Ecol. Indic.* 118, 106698
59. Kim, S.W. *et al.* (2020) Size-dependent effects of polystyrene plastic particles on the nematode *Caenorhabditis elegans* as related to soil physicochemical properties. *Environ. Pollut.* 258, 113740
60. Ju, H. *et al.* (2019) Effects of polyethylene microplastics on the gut microbial community, reproduction and avoidance behaviors of the soil springtail, *Folsomia candida*. *Environ. Pollut.* 247, 890–897
61. Kokalj, A.J. *et al.* (2018) Plastic bag and facial cleanser derived microplastic do not affect feeding behaviour and energy reserves of terrestrial isopods. *Sci. Total Environ.* 615, 761–766
62. Jiang, X. *et al.* (2024) Foliar exposure of deuterium stable isotope-labeled nanoplastics to lettuce: Quantitative determination of foliar uptake, transport, and trophic transfer in a terrestrial food chain. *Environ. Sci. Technol.* 58, 15438–15449
63. Monikh, F.A. *et al.* (2022) Quantifying the trophic transfer of submicron plastics in an assembled food chain. *Nano Today* 46, 101611
64. Deng, Y. *et al.* (2021) Microplastic polystyrene ingestion promotes the susceptibility of honeybee to viral infection. *Environ. Sci. Technol.* 55, 11680–11692
65. Carlin, J. *et al.* (2020) Microplastic accumulation in the gastrointestinal tracts in birds of prey in central Florida, USA. *Environ. Pollut.* 264, 114633
66. Nessi, A. *et al.* (2022) Microplastic contamination in terrestrial ecosystems: a study using barn owl (*Tyto alba*) pellets. *Chemosphere* 308, 136281
67. Cui, Y. *et al.* (2022) Transmission of nanoplastics from *Culex quinquefasciatus* to *Pardosa pseudoannulata* and its impact on predators. *Sci. Total Environ.* 820, 153331
68. Tian, H. *et al.* (2019) Are nanoparticles a threat to mycorrhizal and rhizobial symbioses? A critical review. *Front. Microbiol.* 10, 1660
69. Li, H.H. *et al.* (2024) Arbuscular mycorrhizal fungus alleviates charged nanoplastic stress in host plants via enhanced defense-related gene expressions and hyphal capture. *Environ. Sci. Technol.* 58, 6258–6273
70. Bouaicha, O. *et al.* (2022) Microplastics make their way into the soil and rhizosphere: a review of the ecological consequences. *Rhizosphere* 22, 100542
71. Lu, S. *et al.* (2020) Prevalence of microplastics in animal-based traditional medicinal materials: widespread pollution in terrestrial environments. *Sci. Total Environ.* 709, 136214
72. Zhu, D. *et al.* (2018) Trophic predator–prey relationships promote transport of microplastics compared with the single *Hypoaspis aculeifer* and *Folsomia candida*. *Environ. Pollut.* 235, 150–154
73. Kim, S.W. and An, Y.-J. (2020) Edible size of polyethylene microplastics and their effects on springtail behavior. *Environ. Pollut.* 266, 115255
74. Neher, D.A. and Barbercheck, M.E. (2019) Soil microarthropods and soil health: intersection of decomposition and pest suppression in agroecosystems. *Insects* 10, 414
75. Wardle, D.A. *et al.* (2004) Ecological linkages between aboveground and belowground biota. *Science* 304, 1629–1633
76. Huerta Lwanga, E. *et al.* (2017) Field evidence for transfer of plastic debris along a terrestrial food chain. *Sci. Rep.* 7, 14071
77. Yıldız, D. *et al.* (2022) Effects of a microplastic mixture differ across trophic levels and taxa in a freshwater food web: In situ mesocosm experiment. *Sci. Total Environ.* 836, 155407
78. Al-Jaibachi, R. *et al.* (2018) Up and away: ontogenic transference as a pathway for aerial dispersal of microplastics. *Biol. Lett.* 14, 20180479
79. Wang, Y. *et al.* (2024) From insects to mammals! Tissue accumulation and transgenerational transfer of micro/nano-plastics through the food chain. *J. Hazard. Mater.* 480, 136424
80. Rillig, M.C. (2018) Microplastic disguising as soil carbon storage. *Environ. Sci. Technol.* 52, 6079–6080
81. Miao, L. *et al.* (2019) Acute effects of nanoplastics and microplastics on periphytic biofilms depending on particle size, concentration and surface modification. *Environ. Pollut.* 255, 113300
82. Zhang, Y. *et al.* (2022) Effects of microplastics on soil carbon dioxide emissions and the microbial functional genes involved in organic carbon decomposition in agricultural soil. *Sci. Total Environ.* 806, 150714
83. Shen, M. *et al.* (2023) Recent advances in the research on effects of micro/nanoplastics on carbon conversion and carbon cycle: A review. *J. Environ. Manag.* 334, 117529

84. Rillig, M.C. *et al.* (2021) Microplastic effects on carbon cycling processes in soils. *PLoS Biol.* 19, 3001130
85. Shen, M. *et al.* (2021) Recent advances in impacts of microplastics on nitrogen cycling in the environment: a review. *Sci. Total Environ.* 815, 152740
86. Yin, M. *et al.* (2023) Effects of microplastics on nitrogen and phosphorus cycles and microbial communities in sediments. *Environ. Pollut.* 318, 120852
87. Rong, L. *et al.* (2021) LDPE microplastics affect soil microbial communities and nitrogen cycling. *Sci. Total Environ.* 773, 145640
88. Sun, X. *et al.* (2023) Role of polyamide microplastic in altering microbial consortium and carbon and nitrogen cycles in a simulated agricultural soil microcosm. *Chemosphere* 312, 137155
89. Huang, Y. *et al.* (2019) LDPE microplastic films alter microbial community composition and enzymatic activities in soil. *Environ. Pollut.* 254, 112983
90. Fei, Y. *et al.* (2020) Response of soil enzyme activities and bacterial communities to the accumulation of microplastics in an acid cropped soil. *Sci. Total Environ.* 707, 135634
91. Zhou, Z. *et al.* (2023) Polyethylene microplastic and soil nitrogen dynamics: unraveling the links between functional genes, microbial communities, and transformation processes. *J. Hazard. Mater.* 458, 131857
92. Shi, S. *et al.* (2016) The interconnected rhizosphere: high network complexity dominates rhizosphere assemblages. *Ecol. Lett.* 19, 926–936
93. Leifheit, E.F. *et al.* (2021) Potential effects of microplastic on arbuscular mycorrhizal fungi. *Front. Plant Sci.* 12, 626709
94. Etesami, H. *et al.* (2021) Contribution of arbuscular mycorrhizal fungi, phosphate-solubilizing bacteria, and silicon to p uptake by plant. *Front. Plant Sci.* 12, 699618
95. Wan, L. *et al.* (2023) Global meta-analysis reveals differential effects of microplastics on soil ecosystem. *Sci. Total Environ.* 867, 161403
96. Khalid, N. *et al.* (2020) Microplastics could be a threat to plants in terrestrial systems directly or indirectly. *Environ. Pollut.* 267, 115653
97. Green, D.S. *et al.* (2022) The ecological impacts of discarded cigarette butts. *Trends Ecol. Evol.* 37, 183–192