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PERSPECTIVE

# Perspectives on ecological risks of microplastics and phthalate acid esters in crop production systems

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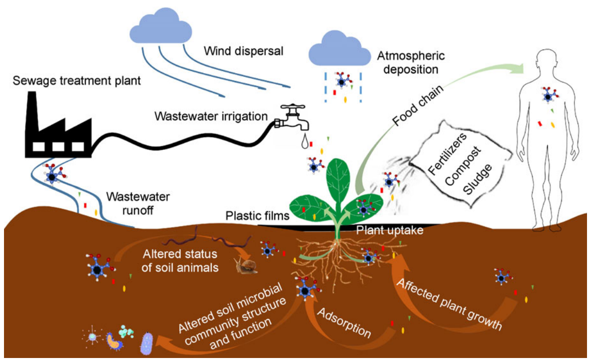
## HIGHLIGHTS

• Microplastics and phthalate acid esters concentrations are positively correlated in soils.
• Phthalate acid esters levels are greatest in Chinese soils.
• Microplastics and phthalate acid esters share common sources and sinks.
• Microplastics and phthalate acid esters are taken up by plants.
• Microplastics and phthalate acid esters exert confounded influences on soil ecosystems.

## GRAPHICAL ABSTRACT

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## ABSTRACT

Microplastics (MPs) and phthalate acid esters (PAEs) co-occur as emerging contaminants of global importance. Their abundance in soil is of increasing concern as plastic-intensive practices continue. Mulching with plastic films, inclusion in fertilizers, composts, sludge application, and wastewater irrigation are all major and common sources of MPs and PAEs in soil. Here, we review studies on the concentration and effects of MPs and PAEs in soil. While there is limited research on the interactions between MPs and PAEs in agroecosystems, there is evidence to suggest they could mutually affect soil ecology and plant growth. Therefore, we propose new research into 1) establishing an efficient, accurate, and simple method to quantify different types of microplastics in soils and plants; 2) exploring the behavior and understanding the mechanisms of co-transfer, transformation, and interactions with soil biota (especially in vegetable production systems); 3) assessing the risk and consequences of combined and discreet impacts of MPs and PAEs on plants and soil biota, and 4) preventing or reducing the transfer of MPs and PAEs into- and within- the food chain.



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## 1 Introduction

As a result of wide application of plastics, both microplastics (MPs) and phthalate acid esters (PAEs) are emerging as troublesome environmental contaminants. PAEs are commonly included in plastics owing to their function as ‘plasticizers’ (Staples et al., 1997), while MPs are defined by their plastic material particle size being less than 5 mm (Qi et al., 2020). As PAEs are not covalently bonded with plastic polymers, there is potential for leaching, migration, and subsequently enhanced abrasion of the plastics containing them (Steinmetz et al., 2016). Accordingly, plastics abrade into progressively smaller particles, including MPs and nanoplastics (NPs) (Duis and Coors, 2016). PAEs and MPs share some common origins as ‘partners’ in plastic polymers used in horticulture and farming. Some of these products are clearly visible on farmlands (plastic films, nets, green packaging from agronomic products, and shredded compost contamination), but there are invisible sources, for example, fragments from fertilizers or colloidal suspensions in irrigation. Nonetheless, the soil receives both the obvious and more ambiguous inputs: being the common sink for both MPs and PAEs.

PAEs (Gu et al., 2017) and MPs (Lv et al., 2018; Qi et al., 2020) have been widely studied as separate contaminants in farmland soil. However, the PAE content of MPs is often not reported, and the affinity and interactions between MPs and PAEs in the soil have become a subject that cannot be ignored. A more comprehensive understanding requires us to elucidate the separate effects of MPs and the PAEs they contain. Some have monitored the effect of MPs on PAE migration (Gao et al., 2019; Li et al., 2020). However, as MPs are also a source of PAEs, it is necessary to study the interacting ‘co-behaviors’ to be able to more confidently allocate ecological risks of MPs and associated PAEs in soil. The present review focuses on the recent progress in characterizing this pollution, behavior, and the entwined ecological effects of PAEs and MPs in soil.

## 2 Occurrence and distribution of PAEs and MPs in soil

In recent decades, the PAE concentration of agricultural soil had been widely investigated (Fig. 1). The average concentrations of PAEs in the agricultural soil in China were higher than those elsewhere (Table 1). Di(2-Ethylhexyl)Phthalate (DEHP) was the most abundant phthalate in all soils, whereas di-butyl phthalate (DBP) was the second most abundant (Hu et al., 2003). There is currently no standard approach for the quantification of plastics in soil, with the separation of MPs from soils being the most difficult and most crucial step. At present, the separation methods include screening, density separation, pressurized fluid extraction, and digestion (He et al., 2018). Methods for the extraction of polypropylene microplastics in swine manure have also been proposed (Wu et al., 2020). The main qualitative means of analyzing MPs is attenuated total reflection-Fourier transform infrared spectroscopy (Weithmann et al., 2018). Due to the emerging concern of MPs in aquatic systems, coastal beach sediments in many coastal areas have been well studied. Although the occurrence of MPs in terrestrial ecosystems has been repeatedly reported, there are significant knowledge gaps. In agricultural soils, MPs with sizes less than 0.5 mm are the most abundant (Table 2). However, it is also difficult to quantify MPs of much smaller size, especially nanoplastics (NPs) with a size range of 1 to 100 nm.

Environmental concentrations of PAEs and abundances of MPs are closely linked, as found in surface seawater from Jiaozhou Bay (Liu et al., 2020b), coastal psammitic sediments in the tropical Atlantic Ocean (Benson and Fred-Ahmadu, 2020), north China (Zhang et al., 2018), and the urban channel of the Ria and coast of Campeche in Mexico (Vered et al., 2019). The concentration of DEHP increased with the abundance of MPs in wastewater ($R^2$ = 0.97, $P < 0.05$) (Takdastan et al., 2021). PAEs in the MPs from sandy beaches of tropical Atlantic ecosystems in Nigeria measured up to 45.6 mg DEHP $kg^{-1}$ plastic (Fred-Ahmadu et al., 2020). PAEs in plastics were even used as tracers to determine the sources of MPs in road dust (Kitahara and Nakata, 2020). However, Li et al. (2021) suggested that environmental factors must be considered to understand the genuinely dynamic relationship between PAEs and MPs in agricultural soil. While MP concentrations depend mainly on input levels, PAE concentrations in soil are more influenced by degradation and dispersal.

## 3 Sources of PAEs and MPs in soil

There are many shared routes through which PAEs and MPs can enter the soil environment, especially for agricultural soils, including plastic mulching, soil amendment with fertilizers, contaminated composts and sewage sludge, irrigation, and atmospheric deposition. In the agricultural context, plastic film was considered the main source of PAEs (Lv et al., 2018) and MPs (Zhou et al., 2020b), followed by fertilizers, composts, and sludges. In gardens and domestic food production, organic fertilizers from biowaste fermentation (anaerobic digestate) and composting are a significant but often overlooked source of microplastics (Weithmann et al., 2018). Irrigation is an essential source of PAEs (Zhang et al., 2015b) and MPs (Ding et al., 2021) for soil in wastewater irrigation areas. Atmospheric deposition introduces both PAEs (Zeng et al., 2010) and MPs (Zhang et al., 2019d) into remote areas.

### 3.1 Plastic film

Plastic films have been used to mulch soil in recent decades worldwide, bringing significant short-term economic benefits.

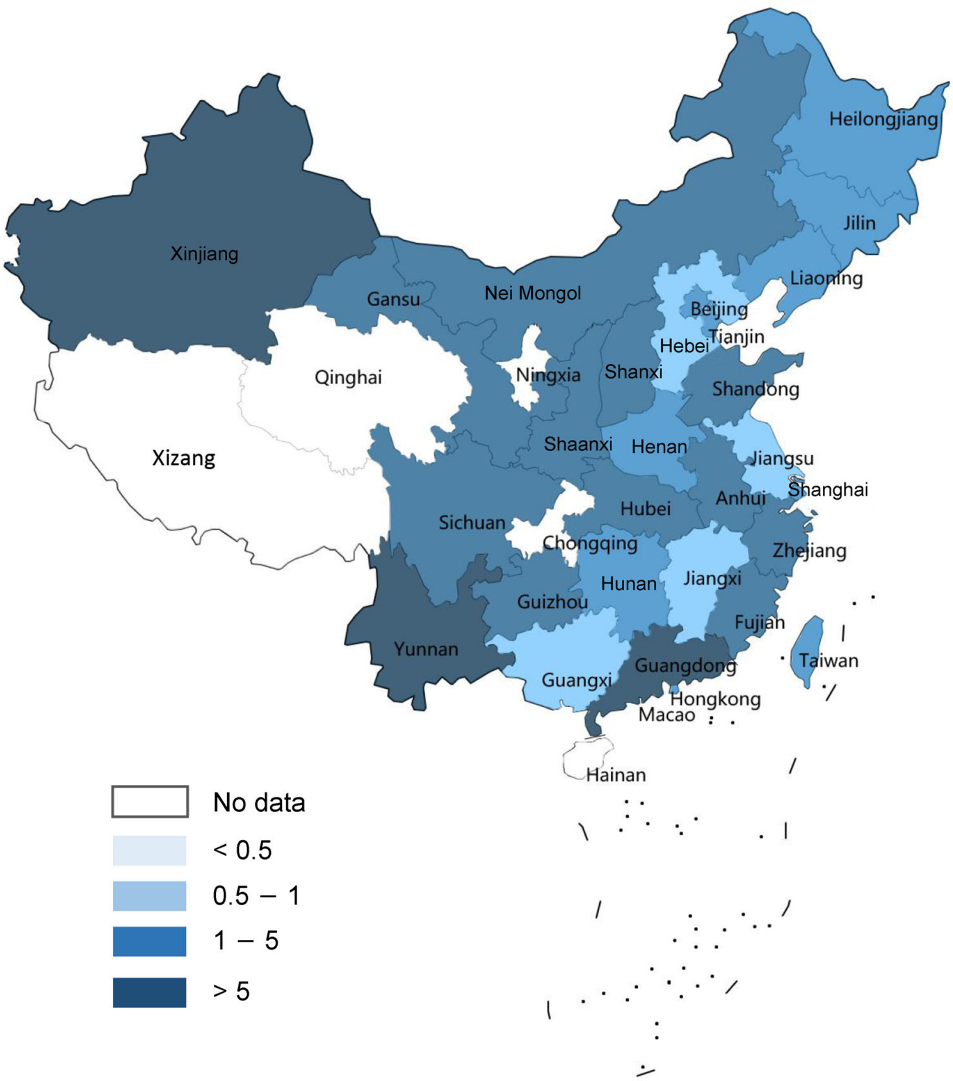


**Fig. 1** Average concentrations of phthalate acid esters (PAEs) in soil in provinces in China .

With the extensive use of plastic mulch, PAEs can migrate directly from films to the soil, but substantial amounts of PAEs are found resident in discarded plastic residues incorporated into the soil, which progressively abrade to MPs and NPs. The average residual rate of plastic mulch was 19.7% following the bulletin of the first national census on pollution sources in China (Lv et al., 2018). Such intensive application and poor disposal of plastic mulch are essential contributors for MPs in terrestrial environments (Table 2). A significant linear correlation was observed between the amount of film mulching and plastic residue in soils (Zhou et al., 2020a). Meanwhile, there was a positive correlation between MPs and PAEs in the greenhouse soil in Xuzhou, China (Li et al., 2021). Timely recovery of plastic film waste is the most efficient (medium to long-term) and effective way to reduce the pollution burden of MPs and PAEs in soil.

## 3.2 Fertilisers, compost, and sludge

The use of organic fertilizers, compost, and sludge in plant production systems (Lowman et al., 2013) is central to maintaining nutrient-use efficiency and soil health through carbon metabolism and ecology of the soil microbiome (Kibblewhite et al., 2008). However, some fertilizers were found to contain quantities of PAEs ranging from 0.01 to 2.8 mg $kg^{-1}$ (Mo et al., 2008), presumably as PAEs leached from plastic packaging. A positive correlation was observed between zfertilizer application rate and PAE content in the soils of Guangdong's greenhouses (Cai et al., 2005; Cai et al., 2008a). The application of composted manure adds extraneous dissolved organic matter (DOM) to the soil, and DOM enhances the adsorption capacity of DBP by soils through partitioning (Wu et al., 2018). Organic fertilizers, manures, composts, and biowaste fermentation (anaerobic digestate)

**Table 1** Average concentrations of phthalate acid esters (PAEs) in soil in China and other countries (mg kg$^{-1}$).

| Country | Soil type | PAE concentration (mg kg$^{-1}$) | | | Reference |
|---|---|---|---|---|---|
| | | Min | Max | Mean | |
| China | Agricultural | 0.032 | 6.29 | 1.09 | Niu et al., 2014 |
| Denmark | Agricultural | 0.012 | 1.9 | 0.39 | Vikelsoe et al., 2002 |
| Scotland | Surface | 0.025 | 1.6 | 0.22 | Rhind et al., 2013 |
| Netherlands | Agricultural | ND | - | 0.032 | Peijnenburg et al., 2006 |
| Serbia | Surface | 0.19 | 2.12 | 0.83 | Skrbic et al., 2016 |
| UK | Agricultural | 0.042 | 0.099 | 0.071 | Gibson et al., 2005 |
| Czech Republic | Agricultural | 0.21 | 3.47 | - | Dankova et al., 2016 |

"-" means no data.

are all known sources of MPs and PAEs in the environment; 16 PAEs were identified in sludge from Shanghai ranging from 22.6 to 1350 mg PAEs kg$^{-1}$ sludge (dry weight) (Zhao et al., 2014). MP content was up to 41 particles g$^{-1}$ in sludge from fields that underwent sludge applications (Corradini et al., 2019). Braun et al. (2021) estimated that typical compost applications drew in plastic loadings of between 84 000 to 1 610 000 plastic items ha$^{-1}$ per year, equivalent to 0.34 to 47.53 kg plastic. Sludge tends to be destined as a soil amendment in most countries (Gao et al., 2020a). Investigations of MPs in sludge amended soils showed a close correlation to the application rate of sludge-based fertilizers (Zhang et al., 2020). Limits and standards for MP and PAE inputs from fertilizers, composts, and sludges will be central to controlling and managing this growing issue.

### 3.3 Wastewater irrigation

With shortages in water resources, wastewater and reclaimed water have been widely used to support agricultural production, especially in areas around industrial cities (Hamilton et al., 2007). Six priority PAEs were found to be ubiquitous environmental contaminants in topsoil obtained from such areas (Zhang et al., 2015b). The amount of treated wastewater used for irrigation is increasing worldwide (Sato et al., 2013). The abundance of MPs in water was up to 9.0 items L$^{-1}$ in the Al-Asfar and Al-Hubail lake (Picó et al., 2020), with the abundance of MPs in soil being affected by the type of irrigation water (Wang et al., 2020b). During wastewater treatments, the addition of polymeric flocculants has been considered a potential source (Hurley and Nizzetto, 2018).

### 3.4 Atmospheric deposition

Atmospheric PAE deposition and particulate deposition fluxes were closely related in the subtropical city of Guangzhou, China (Zeng et al., 2010). Atmospheric transport is also an essential pathway for the transport of MPs from industrial to remote areas (Zhang et al., 2019d). In the Pyrenean Mountains, deposition of MPs was attributed to cities, agricultural, and industrial areas (Zhang et al., 2019d). Thus PAEs in the soil and atmosphere are distributed, migrate, redistribute, and remigrate toward equilibrium (Gu et al., 2017).

To more accurately describe the movement and impacts of both classes of contaminants in the environment requires an understanding of the affinity between them, i.e., the factors determining their partnership and departure from one another. While, for the most part, MPs as physical particles can be seen to move unaffected by their PAE content, there are some exceptions where the PAE content may affect the movement of MPs. This is because PAEs modify the physical properties of a plastic, for example, improving 'plasticity' or resilience against abrasion. Therefore, the PAE content (related to age) may well affect the abrasion and shedding of these MPs into the environment. Perhaps the best example can be found in plastic recreational surfaces. Plastic grass or 'artificial lawns' are an increasingly popular way of simulating a lush ecosystem in gardens and sports venues (Francis, 2018), but the environmental impacts of abraded dust from cleaning and wear are still not well understood (Francis, 2018). Artificial lawns can include PA, or PE-based materials, sewn into an expanded polypropylene mesh, sometimes combined with butyl rubber or latex linings, and were previously found to host a range of plasticizers and PAEs (Celeiro et al., 2018). Over time, these finely woven plastics are increasingly vulnerable to physical wear under foot traffic, resulting in the release of MPs and their associated PAEs into the atmosphere. While initially resilient, we expect the abrasion of these surfaces to accelerate as plasticizers are lost from the matrix.

## 4 Behavior of PAEs and MPs in soil

The behavior of PAEs in the soil is reasonably well documented. For example, Wu et al. (2018) found that the equilibrium adsorption quantity for DBP was positively related to the soil organic matter content, and hydrophobic partition-

**Table 2** Abundance of microplastics (MPs) in the soil. Plastics commonly reported as polypropylene (PP), polyethylene (PE), polystyrene (PS), and polyamides (PA).

| | Soil type | Samples number | Size (mm) | Abundance (pieces $kg^{-1}$) | | Main forms | Source | Main material | Main size (mm) | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Mean | Range | | | | | |
| Yeoju City, Korea | Agriculture soil | 68 | < 5 | 664 | - | - | Plastic film | PE, PP | < 1 (41.5%) | Choi et al., 2020 |
| Chile | Agriculture soil | 31 | < 5 | - | 1100–3500 | Fiber (97%) | Sewage, Sludge | - | < 0.05 | Corradini et al., 2019 |
| Oaxaca, Mexico | Pasture soils | | < 5 | - | 1490–1530 | Films, Pellets | Deposition | - | 0.15–0.5 | Alvarez-Lopez-tello et al., 2021 |
| Shanghai, China | Agriculture soil | 20 | 0.02–5 | - | 62.5–78.0 | PP (50.5%), PE (43.4%) | Plastic film, Compost | - | - | Wu et al., 2018a |
| Shouguang, China | Agriculture soil | 135 | < 5 | 1444±986 | 310–5698 | Fragment, Film | Plastic film, Flooding | PP, PE | < 0.5 (65.2%) | Yu et al., 2021 |
| Shouguang and Xuzhou, China | Agriculture soil | 34 | < 5 | - | 380–3786 | - | Plastic film | - | - | Li et al., 2021 |
| Wuhan, China | Agriculture soil | 20 | 0.02–5 | - | 320–12560 | Fiber, Microbead | Wastes, Sewage | PA (32.5%) PP (28.8%) | < 0.2 (70%) | Chen et al., 2020 |
| Inner Mongolia, China | Agriculture soil | 6 | < 5 | - | 2526–6070 | Film (38.6%) | Plastic film | - | < 1 | Wang et al., 2020b |
| Hangzhou Bay, east, China | Agriculture soil | 60 | < 5 | - | 263–571 | Fragment, Fiber | Plastic film, Irrigation, Waste, Compost | - | 1–3 | Zhou et al., 2020a |
| Yunnan, China | Agriculture soil | 100 | < 5 | 9800 | 900–40800 | Fragment (78.3%) | Plastic film | - | < 0.5 (89.3%) | Huang et al., 2021 |
| Shaanxi, China | Agriculture soil | 9 | < 5 | - | 1430–3410 | Fibers, Particles | Irrigation, Sewage, Sludge | - | 0–0.49 | Ding et al., 2020 |
| Heilongjiang, China | Mollisol farmlands | 88 | < 5 | - | 100±100–400±692 | - | - | PE | - | Zhang et al., 2019c |
| Jiangxi, China | No fertiliser applied soil | 2 | < 5 | 16.4±2.7 | - | - | - | - | - | Yang et al., 2020 |
| | Fertiliser applied soil | 10 | < 5 | 43.8±16.2 | | - | Organic manures | | | |

(*Continued*)

| | Soil type | Samples number | Size (mm) | Abundance (pieces $kg^{-1}$) | | Main forms | Source | Main material | Main size (mm) | Reference |
|---|---|---|---|---|---|---|---|---|---|---|
| | | | | Mean | Range | | | | | |
| Shanghai, China | Rice-fish coculture | - | - | 10.3±2.2 | - | Fiber | Organic fertiliser, Fish diets | PE, PP | < 1 | Lv et al., 2019 |
| Switzerland | Floodplain soils | 29 | < 5 | 593 | - | - | Diffuse aeolian transport | - | - | Michael et al., 2018 |
| South-western, China | Forest buffer zone | 55 | < 10 | 18760 | 7100–42960 | Fiber (92%) | Sewage, Sludge, Wastewater irrigation | - | 0.05–1 (95%) | Zhang and Liu, 2018 |
| North-west, China | Sand<br>Grassland<br>Woodland | 26 | < 5 | 2696.5 | 1360–4960 | Fiber (39%) | Plastic products | - | < 0.5 (88%) | Ding et al., 2021 |

“-” means no data.

ing played a dominant role in the DBP adsorption process. However, the interaction effects of MP identity and MP particle size on the adsorption of PAEs are still unknown. While PAEs can be dispersed into the environment independently of MPs, the movement of MPs and PAEs through the environment are coupled in several ways, as they share an affinity for one another. The first aspect of this affinity is the straightforward inclusion of PAEs during the manufacture of the original plastics and subsequent dispersal of the PAEs contained within as microplastics shed and abrade into the environment: in this respect, MPs function as a source of PAEs. However, the second aspect of this affinity operates in the reverse direction, where PAEs are adsorbed by microplastics.

### 4.1 Adsorption of PAEs on MPs

Due to their small size and large specific surface area, MPs are capable of adsorbing large quantities of persistent organic pollutants (POPs) (Rios et al., 2007), including PAEs. Different adsorption mechanisms exist between organic pollutants and microplastics, depending on the chemistry of MP and PAE, respectively. In aqueous ecosystems, the sorption isotherms of diethyl phthalate (DEP) and di-butyl phthalate (DBP) on MPs (PS, PVC, PE) were highly linear. The higher adsorption rates for DBP relative to DEP and greater hydrophobicity of DBP again suggested that hydrophobic interactions are of major importance (Liu et al., 2019). Here, the solution pH and natural organic matter content had no significant impact on the adsorption of PAEs to MPs. However, due to the presence of the salting-out effect, the adsorption of PAEs on MPs was enhanced by NaCl and $CaCl_2$ (Zhang et al., 2019a).

### 4.2 Uptake by plants

A positive correlation was observed between contamination level of soil and plant-uptake of PAEs (Gu et al., 2017). The pathways of absorption are as follows: (1) Plant roots absorb PAEs directly from the soil solution and transfer via the xylem to leaves by transpiration flow, with PAEs accumulating in the edible parts, as was found with corn and soybeans (Chen et al., 2005; Cai et al., 2008b; Sun et al., 2010); (2) above-ground biomass can absorb PAEs directly from the air. With leafy vegetables, such as Chinese cabbage, this can be especially problematic (Zeng et al., 2007). MPs are also understood to be capable of entering into edible biomass. Polystyrene and polymethylmethacrylate particles with submicron and nanometre sizes were able to penetrate the stele at sites of lateral root emergence via ‘crack-entry mode’ followed by transport to shoots (Li et al., 2020). High transpiration rates promoted the uptake of plastic particles, indicating that transpiration pull was the main driving force of their movement (Li et al., 2020). However, we know of no studies on the co-migration of PAEs and MPs from soil to plant biomass. The interactions in flow and transfer of MPs and PAEs in soil-plant systems are thus still unclear and need to be explored if we are to gain control of these phenomena. PAEs may first adsorb onto MPs which may or may not facilitate entry. In addition, PAEs already present in submicron MPs may escape the MPs and relocate/accumulate with an expected affinity for plant lipids.

### 4.3 Degradation of PAEs and MPs

As the main process of removing POPs, microorganisms play a major role in PAE degradation in the environment under various conditions (Staples et al., 1997). As a new habitat for bacterial growth, MPs could affect the composition of the bacterial community in environmental media (Chi et al., 2021). Laboratory experiments show that the addition of microplastics with a higher affinity for DBP inhibited the degradation of DBP in sediments (Chi et al., 2021). The effect of MPs on the degradation of PAEs may be related to other characteristics, which need further investigation.

Tillage, digging, bioturbation will all increase specific surface area. Old and degraded plastics eventually become fragile and decompose into smaller and smaller microparticles, becoming microplastics. Mechanical erosion of plastic surfaces led to delamination of the surface then produced microplastic particles (Browne et al., 2007; Barnes et al., 2009). As PAEs are not covalently bonded into the polymer structure, the fragmentation of macroplastics may accelerate the release of PAEs and MP/NP-facilitated transport.

## 5 Impact of PAEs and MPs on soil ecosystems

Ecotoxicological studies on PAEs or MPs have focused primarily on the simple deleterious effects of one or the other on plants, fauna, and microorganisms. However, a more accurate description of the consequences of co-pollution requires a multidirectional approach. This includes understanding the factors governing the migration of microplastics and PAEs in biological context: which requires an understanding of biological vectors, especially changes in the behavior of influential soil organisms.

### 5.1 Soil macrofauna

A dose-response relationship between DNA damage in earthworm cells and exposure concentrations of PAEs was studied. With the extension of exposure time, PAEs can cause a decrease in foraging and metabolism, weight loss, swelling, congestion, and even ulceration of earthworms (Wang et al., 2010). The biota-to-soil accumulation factors of DEHP in earthworms (*Eisenia fetida*) were 0.17 and 0.07 in agricultural and forest soil, respectively (Hu et al., 2005). Earthworm (Prendergast-Miller et al., 2019) burrowing, feeding, and cast behavior (which governs the distribution of many substances and associated biota) were found to be significantly affected by ingestion of polyester MP fibers (Prendergast-Miller et al., 2019). Effects such as these will impact upon a range of critical soil properties, with cascading consequences. However, it is not known if the effects were due specifically to the physical MPs themselves or MP-resident PAEs. Other studies of the same model earthworm (*Lumbricus terrestris* L.) found MP particle-size selectivity for polyester (Lwanga et al., 2016). Soil fauna facilitated the horizontal and vertical transport of MPs in the work of Xu et al. (2020). A simulated experiment confirmed that exposure to high levels of PE MPs (>28%, not environmentally realistic) inhibited the growth and increased earthworm fatality, while the reproduction rate was unaffected (Zhu et al., 2018). Nonetheless, manual addition of more realistic quantities of low-density polyethylene fragments in the field significantly affected the composition and decreased the abundance of microarthropod and nematode communities (Lin et al., 2020). PAEs and MPs may have synergistic effects causing damage to soil fauna, and the PAE content may be the more influential component explaining the observed impacts of MPs.

### 5.2 Soil microorganisms

The effect of PAEs on microbial community in soils depends on the specific identity and concentration of the PAE. Dimethyl phthalate (DMP) or DEP with the concentrations of $\leqslant 50$ mg kg$^{-1}$ soil showed little effect on the microbial activity, whereas concentrations $\geqslant 100$ mg kg$^{-1}$ soil had statistically significant and deleterious effects (Chen et al., 2013). DOP (Chen et al., 2013) and DEHP (Cartwright et al., 2000) do not exhibit any significant impact on soil microbial activity; even at 100 mg kg$^{-1}$, DEHP had no effect on membrane fluidity (Cartwright et al., 2000). DBP was found to inhibit the activity, diversity, and heterogeneity of microorganisms in the soil, and also increased microbial utilization of amines and carboxylic acids and decreased the utilization of carbohydrates and amino acids (Gao et al., 2020b).

With the accumulation of MPs in the environment, MPs may gradually affect the bacterial community in environmental media (Julia et al., 2018). MPs in agricultural soils were found to accumulate colonizing microorganisms (Zhang et al., 2019b), with MPs possibly serving a role in the attraction of hydrophobic C sources (Hirai et al., 2011). Additions of MPs have been found to significantly affect soil microbial respiration and activity of soil b-glucosidase, urease, and phosphatase (Yang et al., 2018). In the study of Gao et al. (2020a), MP addition promoted soil carbon dioxide emissions, and even accelerated succession of the soil bacterial community (Wang et al., 2020a). We infer that PAEs and MPs may have antagonistic effects on the abundance and diversity of a range of microorganisms: except for those responsible for metabolizing organic compounds with an affinity for microplastics. Considering the increasing abundance of MPs and PAEs in soils, these trends may continue.

### 5.3 Plants

The migration of pollutants in soil-plant systems is of paramount importance for food quality, and human health. Accumulation of PAEs in plants affects germination, growth and development as well as interfering with plant metabolism- which in turn affects pigmentation, osmolytes, stress biomarkers and activities of antioxidative enzymes– thus reducing the edible yield and quality (Kumari and Kaur, 2020). In this instance, DBP accumulation decreased soluble protein content and increasing nitrate content of *Brassica napus* (Kong et al., 2018), with DBP again emerging more toxic than DEHP (Ma et al., 2018).

Direct toxicity and indirect effects (altering bioturbation, soil structure, nutrient immobilization, transporting or adsorbing contaminants) were considered the main mechanisms for microplastic effects on plants (Rillig et al., 2019). MPs of 100 nm strongly inhibited seedling establishment and growth of *Arabidopsis thaliana* (Sun et al., 2020), but the joint toxicity of PAEs and MPs still needs disentangling. In hydroponic systems, the addition of exogenous MPs increased growth inhibition by DBP in lettuce by aggravating photosynthetic activity (Gao et al., 2019). DBP and polystyrene MPs could also disturb microalgal growth. When the concentration of MPs was less than 10 mg L$^{-1}$, low concentrations of DBP were antagonistic to MPs inhibition of microalgal growth, while a high concentration of DBP enhanced the negative impact of MPs (Li et al., 2020).

Both PAEs and MPs can negatively impact plant growth. The accumulation of MPs in the edible parts of plants increases the risks of human intake of sorbed POPs, and especially PAEs. Risk assessment of PAEs for multiple exposure routes has been established, including inhalation and dermal contact (Wang et al., 2021), and oral intake (Wang et al., 2015). However, the current understanding of the potential impacts on human health associated with an abundance of MPs is limited and insufficient to accurately assess the health risks of consuming them.

## 6 Conclusions and perspective

MPs comprise the crushed debris of various plastic polymers and can both be a source or a sink of PAEs. While 'straight' PAE effects have been widely studied in soils, the interactions with related MP content, soil health, and ecosystem processes need urgent attention. The interactions between PAEs and MPs– including adsorption and degradation– affects both their behavior and persistence in the environment. However, their entwined behavior still lacks sufficiently descriptive data, especially regarding their co-migration and departure from one another. Many challenges remain to be tackled, including the methodologies for extraction and characterization of MPs and PAEs. Specific issues need to be addressed: 1) establishing an efficient, accurate, and simple method to quantify different types of microplastics in soils and plants; 2) exploring the behavior and understanding the mechanisms of co-transfer and transformation as interacting with soil biota (especially in vegetable production systems); 3) assessing the risk and consequences of combined and discreet impacts of MPs and PAEs on plants and soil biota, and 4) preventing or reducing the transfer of MPs and PAEs into - and within - the food chain.

## Conflict of interest

The authors declare no competing financial interest.

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