Particulate plastics-plant interaction in soil and its implications: A review

Published in: *Science of the Total Environment*

Citation for published version: Wu, X., Lu, J., Du, M., Xu, X., Beiyuan, J., Sarkar, B., Bolan, N., Xu, W., Xu, S., Chen, X., Wu, F., Wang, H., (2021) Particulate plastics-plant interaction in soil and its implications: A review. *Science of the Total Environment*. 792: 148337. doi: 10.1016/j.scitotenv.2021.148337.

Document version: Accepted peer-reviewed version.

Particulate plastics-plant interaction in soil and its implications: A review

Xiaolian Wu[a], Jinlian Lu[a], Minghui Du[a], Xiaoya Xu[a], Jingzi Beiyuan[a], Binoy Sarkar[b], Nanthi Bolan[c], Weicheng Xu[a], Song Xu[a], Xin Chen[a], Fengchang Wu [d], Hailong Wang[a,e,*]

[a] *School of Environmental and Chemical Engineering, Foshan University, Foshan, Guangdong 528000, China*

[b] *Lancaster Environment Centre, Lancaster University, Lancaster, LA1 4YQ, United Kingdom*

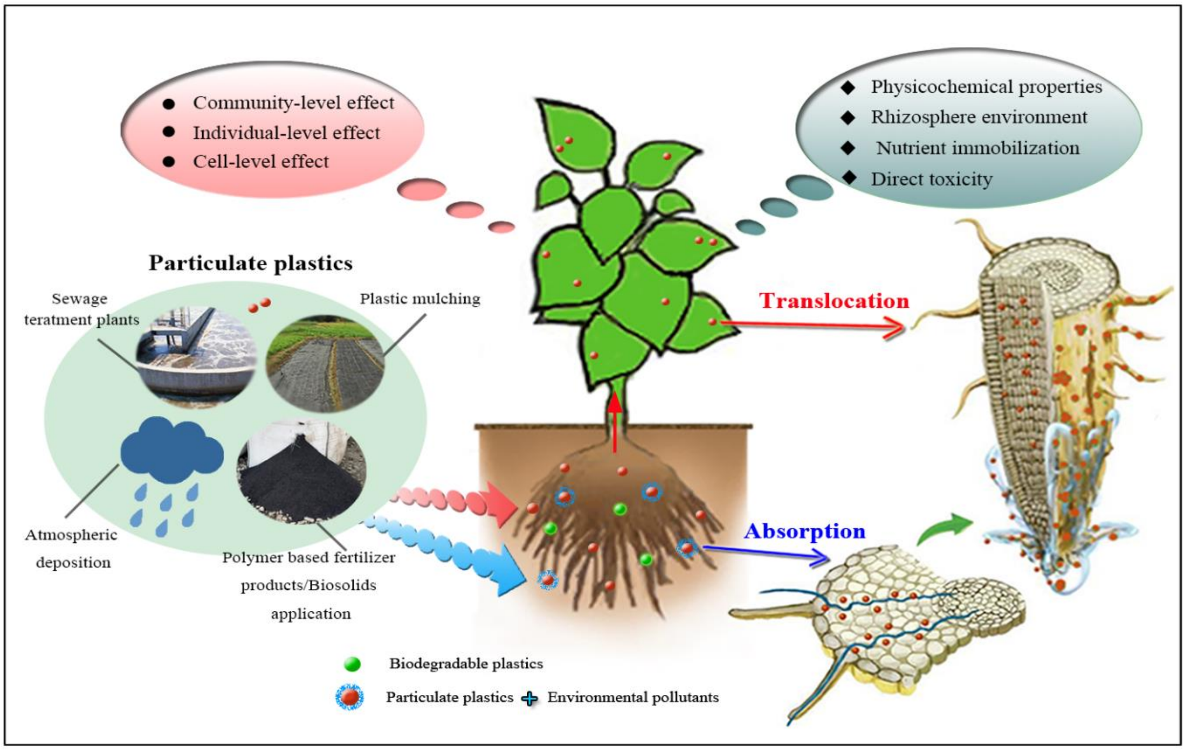
[c] *Global Centre for Environmental Remediation (GCER), School of Engineering, Science and Environment, The University of Newcastle, Callaghan, NSW 2308, Australia*

[d] *State Key Laboratory of Environmental Criteria and Risk Assessment, Chinese Research Academy of Environmental Sciences, Beijing, 100012, China*

[e] *Key Laboratory of Soil Contamination Bioremediation of Zhejiang Province, Zhejiang A&F University, Hangzhou, Zhejiang 311300, China*

*Corresponding author. Hailong Wang, Email: *hailong.wang@fosu.edu.cn*

Graphical abstract



Highlights

- Nanoplastics generated via plastic weathering can be taken up by plants.
- Plastics are taken up via endocytosis, apoplastic transport and crack-entry.
- Plastics may cause community, individual and cell-level effects on plants.
- Plastics significantly alter the bioavailability of environmental pollutants in soil.

ABSTRACT

Particulate plastics (<5 mm), including macroplastics (1 μm to 5 mm), microplastics (100 nm to 1 μm) and nanoplastics (<100 nm), have become a global environmental problem due to their widespread occurrence, distribution and ecosystem risk. Although numerous studies on particulate plastics have been conducted in aquatic systems, investigations in the soil ecosystem are lacking. Soil is the main storage place of particulate plastics, conferring significant impacts on plant growth and development. The impact of particulate plastics on plants is directly related to the safety of agricultural products. This review comprehensively examines the pollution characteristics and exposure pathways of particulate plastics in agricultural soils, highlighting plastic uptake process and mechanisms in plants, and effects of particulate plastics, biodegradable particulate plastics and combined pollution of plastics with other environmental pollutants on plant performances. This review identifies a number of future research prospects, including the development of accurate quantitative methods for plastic analysis in soil and plant samples, understanding the environmental behaviors of conventional and biodegradable particulate plastics in the presence and absence of other environmental pollutants, unravelling the fate of particulate plastics in plants, phyto-toxicity and molecular regulatory mechanisms of particulate plastics, and developing best management practices for the production of safe agricultural products in plastic-contaminated soils.

Keywords: Nanoplastics; Microplastics; Biodegradable plastics; Toxicity; Uptake; Soil contamination.

1 Introduction

Plastics are commonly used in our daily life and majority of the plastic products (approximately 99%) are discharged into the terrestrial environment following their usage (Van Sebille et al., 2015). A recent study showed that the global plastic emissions could reach 53 Mt/year by 2030 (Borrelle et al., 2020), of which around 79% would be landfilled or abandoned in the natural environment (Geyer et al., 2017). The environmental fate, potential impacts and ecological risks and analytical methods of plastics have been extensively studied in aquatic environments in the recent years. However, only a small number of studies focused on the environmental effects of plastics in the terrestrial ecosystem. In the environment, “microplastics” exist in various particle size fractions, including macro- (1 μm to 5 mm), micro- (100 nm to 1μm) and nano-plastics (1 nm to 100 nm) (Wallace, 2016). An alternative phrase “particulate plastics” is also often used to represent the pollution of the environment with plastics of different particle sizes (Bolan et al., 2020).

Particulate plastics can enter farmland soils in large quantities through the use of agricultural films, polymer-based slow release fertilizers, sewage irrigation, compost and biosolids application, atmospheric sedimentation and surface runoff to form a combined pollution with organic and inorganic pollutants in soils (Rillig, 2012; Weithmann et al., 2018; Bolan et al., 2020). Preliminary studies argued that the storage of plastics in the soil could be much more than that in the aquatic system (Nizzetto et al., 2016). Once in the soil, plastic debris would be fragmented to smaller particles, such as microplastic particles (with sizes below 5 mm), by mechanical abrasion, UV exposure and/or biological weathering (Song et al., 2018). Not surprisingly, because of being highly resistant to degradation, particulate plastics are ubiquitous in the soil, and eventually would reach levels that could affect the quality of the soil ecosystem (de Souza Machado et al., 2018;

Kumar et al., 2020). It is a big challenge to investigate the sources, fate and ecological effects of particulate plastics in the terrestrial environment, especially in agricultural soils where plants might also take up some of these tiny particles, resulting in a contamination risk in foods. Fig. 1 illustrates selected reports on particulate plastics in the field of plant science in recent years. Only limited number of studies are currently available on the accumulation of particulate plastics in plants and the subsequent effects on plant physiology (Kumar et al., 2020). Due to the universally recognized ecological risk of particulate plastics to the aquatic environment (Sridharan et al., 2021), it is necessary to investigate the fate and transformation of particulate plastics in agricultural soils, and their entry pathways into the plant body with or without plastic-associated contaminants. This review specifically aims to address the plant uptake of particulate plastics from the soil and subsequent effects on plants and the food chain. The key objectives of this article are to: (1) outline the potential routes of particulate plastics entry into the soil, and the distribution characteristics of particulate plastics in the soil, (2) discuss the particulate plastic contamination in plants and mechanisms of plant uptake; (3) explore the effects of particulate plastics on plants and associated physiological and biochemical mechanisms, and (4) outline the interactions of particulate plastics and other environmental pollutants, and biodegradable particulate plastics with plants.

2 Contamination of particulate plastics in agricultural soils

2.1 Sources of particulate plastics in agricultural soils

Given plastics are widely demanded in people's daily life, and plastic usage and disposal are not yet regulated in most of the countries, tracing the origin of particulate plastics in the soil and unravelling their potential routes to the soil is quite challenging. The soil ecosystem is the most important driver for human food production. Once particulate plastics enter the soil, they are difficult to be degraded, which may result in ecotoxicological effects on soil-based organisms (e.g., plants, earthworms, microbes) (Rillig et al., 2017; Lozano et al., 2020; Zhang et al., 2020a). Therefore, understanding the routes of particulate plastics in the soil is a key to evaluating and characterizing the extent of soil plastic contamination. Based on literature reports, a number of routes for the entry of microplastics into the soil can be postulated, such as application of polymer based slow release fertilizers, composts, biosolids, and sludges (Corradini et al., 2019; Crossman et al., 2020; Zhang et al., 2020a), plastic mulching (Zhang and Liu., 2018; Li et al., 2020a; Huang et al., 2020), waste water irrigation (Sighicelli et al., 2018; Wang et al., 2019), and atmospheric deposition (Liu et al., 2020).

Particulate plastics are universally detected in wastewater treatment plants (WWTPs), and nearly 99% of particulate plastics are removed from wastewater during the treatment (Lee and Kim., 2018), which ultimately accumulate in the sewage sludge (Liu et al., 2018a). Sewage sludge (i.e., biosolids) containing copious amount of particulate plastics are often utilized (approximately 50%) (Nizzetto et al., 2016) as organic soil amendments in many countries (Coors et al., 2016; Crossman et al., 2020). However, this agronomic practice is identified to contribute a major route of particulate plastics into the farmland (Corradini et al., 2019). In China, based on the total sewage sludge applied to soils per year, an average of 22.7 ± 12.1 particles/g particulate plastics are brought into the soil via sewage sludge alone (Li et al., 2018a). In Europe, the total accumulation of particulate plastics in the soil via sewage sludge application was estimated to be 63,000 to 430,000 t/year, which is far more than the total plastic enrichment in global oceanic waters (Nizzetto et al., 2016). However, an accurate estimation of the contribution of sludge-based biowaste products to soil particulate plastic contamination is challenging because most published studies have counted only particles bigger than 1 mm (Weithmann et al., 2018). The lack of detailed studies regarding the size range, shape, and type of particulate plastics present in the soil environment also add to the difficulty to precisely estimate the contribution of sewage sludge to soil microplastic contamination.

Although the removal efficiency of particulate plastics can be high in WWTPs, an abundance of particulate plastics is detected at the outlet of the sewage treatment plant (Lee and Kim., 2018). With a 52% removal efficiency of particulate plastics in WWTP, Kalčíková et al. (2017a) estimated that the global emission of polyethylene (PE) particulate plastics to river could reach 1,125,500,000 particles per day. Besides, contaminated freshwaters, such as lake (Sighicelli et al., 2018) and river (Nizzetto et al., 2016) waters, were also found loaded with high concentration of particulate plastics. For example, a high concentration of particulate plastics was found in freshwater lake from the Yellow River basin of northern China where particulate plastics numbers ranged from 1,760 to 10,120 particles/m^3 (Wang et al., 2019). Irrigation of crops with such contaminated water would undoubtedly lead to the addition of particulate plastics to agricultural soils (Nizzetto et al., 2016).

In addition to sewage sludge discharge, biowastes from fermentation and composting also contribute to microplastic input into agricultural soils. Weithmann et al. (2018) studied the quantitative contribution of particulate plastics into the soil due to biowastes from fermentation and composting, and found that all biowaste samples contained different levels of particulate plastics, bringing in between 35 billion and 2.2 trillion particulate plastics (only counted particles bigger than 1 mm) into the environment each year in Germany alone. Another study showed that the total particulate plastics addition to Canadian agricultural soils via biosolids from WWTPs was up to 3.8×10^9 particles in 2017 (Crossman et al., 2020).

Plastic mulching is a traditional method to enhance crop growth, and more than 128,652 km^2

of agricultural lands are covered with plastic films around the world (Zhang et al., 2019). Following the use of plastic films, the aged debris remains in the agricultural soil (Steinmetz et al., 2016). A successive enrichment of plastic fragments in soils following plastic mulching has been reported in several studies (Ramos et al., 2015; Steinmetz et al., 2016; Saglam et al., 2017; Huang et al., 2020). Zhang et al. (2020a) presented a nationwide projection in China with more than 3600 soil samples, showing that the accumulation of plastic film residues in croplands could reach as high as 550,800 ton. Ramos et al. (2015) found that 3 g PE per m^2 soil was detected in horticultural soils in Argentina, representing for 10% of the area of total sampled soil. However, the contribution of particulate plastics to soils from plastic films in China (Zhang et al., 2018) was smaller than that from soil-applied sludges in Chile (Corradini et al., 2019). It was estimated that the annual contribution of residual particulate plastics to soils from sludges in Chile was nearly 101 times higher than that from plastic film mulching in China (Corradini et al., 2019). Addition of particulate plastics to soils through plastic film mulching and/or polymer-based slow-release fertilizers is closely related to the agronomic practices of the concerned farmlands, such as the frequency and area covered by film mulching, and the type and frequency of fertilizer application (Corradini et al., 2019; Kumar et al., 2020). Nevertheless, the problem of soil particulate plastic pollution caused by plastic films should not be ignored (Zhang et al., 2020a).

Atmospheric transport is another important source of particulate plastics entering into agricultural soils (Liu et al., 2020). In the atmosphere over the city of Paris, about 29-280 particles/m^2 of particulate plastics were deposited each day in 2014 (Dris et al., 2015). Particulate plastics can be transported long distances from contaminated areas to remote areas (Allen et al., 2019). As a result, particulate plastics are ubiquitously present leaving almost no clean agricultural soil in the world (Allen et al., 2019; Kumar et al., 2020). Moreover, particulate plastics in the atmosphere could directly attach on plant leaves, which would greatly interfere with plants' photosynthetic efficiency, and increase the risk of direct contact of particulate plastics with humans (Liu et al., 2020). In the top 11 Green Countries (Chen et al., 2019a), around 0.13 trillion particles of particulate plastics were estimated to be attached to plant surfaces (Liu et al., 2020), suggesting that the deposition of atmospheric particulate plastics has a great contribution to the agricultural system.

Another easily overlooked source of particulate plastic pollution in soils is the use of slow-release fertilizers (Stubenrauch and Ekardt, 2020). Polymer particles are added in traditional mineral fertilizers to scarify the soil and enhance soil's water holding capacity (European Commission, 2017). Moreover, polymers are used for coating fertilizer granules to prevent their caking in the soil (Pietra, 2019). The above polymers might be released from fertilizers into the soil with an increase of the residence time of fertilizers, and may end up accumulating in the soil (Stubenrauch and Ekardt, 2020). According to the European Chemicals Agency (ECHA), banning the use of plastic particles (polymers) as fertilizer additives or coatings could reduce the average annual emission of plastic particles in European soils by approximately 262,500 tonnes within 20 years (European Chemicals Agency, 2019).

2.2 Particulate plastic distribution characteristics

Although soils have been recognized as a major sink of particulate plastics, the distribution characteristics of particulate plastics in soil has been addressed in just a few publications. The concentration of particulate plastics in soil could be a lot higher than that reported in aquatic environments (Fischer et al., 2016; Horton et al., 2017; Zhang and Liu, 2018). The concentration, number, type, and morphology of particulate plastics in the soil are important parameters to assess the extent of particulate plastics pollution in the terrestrial environment. With the development of new analytical methods and deepening of understanding about particulate plastics pollution in the agroecosystem, research works gradually were extended to agricultural soils (e.g., farmland, orchard soil) (Zhou et al., 2019a; Kumar et al., 2020; Sridharan et al., 2021).

Data in **Table 1** shows examples of high concentration of particulate plastics in agricultural soils (Chen et al., 2019b). Zhang and Liu (2018) found that the concentration of particulate plastics in farmland from southwest China was ranging from 7,100 to 42,960 particles/kg. Compared to the Chinese scenarios, a remarkably lower particulate plastic contamination level was found in German farmlands, which contained an average of 0.34 ± 0.36 particles/kg (Piehl et al., 2018). The high levels of particulate plastic contamination in China were related to plastic mulching and biosolid application in soils. The reason for such significant difference between Chinese and German scenarios might be that the clean agricultural soil in Germany would never have received biowastes and agricultural plastic films (Piehl et al., 2018; Harms et al., 2020). Human activity is also an important factor affecting the distribution of particulate plastics in agricultural soils (Chen et al., 2019b). The suburban areas of a city store garbage from the urban areas, concentrating a large number of plastic particles in the peri urban soils and leading to a serious pollution problem. For example, particulate plastics pollution in vegetable soils adjacent to suburban roads in China was about 1.8 times higher than that in residential areas (Chen et al., 2019b).

Many recent studies reported that agricultural soils are polluted with particulate plastics that are mainly less than 1.0 mm in size (Liu et al., 2018b). In southwest China, 82% particulate plastics in agricultural soil samples were in the size range of 0.05 to 0.25 mm (Zhang and Liu, 2018). Particulate plastics of different size distributions and variety of shapes were observed in agricultural soils in the study by Zhou et al. (2019a), who found that the size distribution of most plastic films and fragments (approximately 70%) were <1.0 mm, while most plastic fibers size was in 0.2-0.5 mm range. These results imply that the size distribution of particulate plastics may depend on shapes of the plastic particles. The main shapes of particulate plastics in agricultural soils are fibers, fragments, and films (Zhang and Liu, 2018; Chen et al., 2019b). Polyethylene (PE), polypropylene (PP), polyester (PES), polyethylene terephthalate (PET), polyamide (PA), polyvinyl chloride (PVC), polymethyl methacrylate (PMMA), and polystyrene (PS) are the dominant types of particulate plastics found in agricultural soils (**Table 1**). The distribution characteristics of particulate plastics in agricultural soils are related to many factors, such as the contamination sources, land use practices, soil tillage patterns, soil erosion and so on (Piehl et al., 2018; Chen et al., 2019b; Zhou et al., 2019a).

3 Plant uptake

3.1 Particulate plastics in plants

Particulate plastics are likely to stick strongly on plant root surfaces due to the strong adhesiveness of the polymer particles, and then be taken up into plant roots (Li et al., 2019, 2020b). Nanoscale particulate plastics have similar characteristics to that of nanoparticles, including very large specific surface area. A large number of studies show that plants can take up nanoparticles (Durgesh et al., 2016; Zhang et al., 2016). However, the distribution of nano-/microplastics in plants is poorly understood (Bosker et al., 2018; Li et al., 2019; Sun et al., 2020). Particulate plastics might be absorbed into the plant roots, and then transferred from the roots to stems and leaves via the transpiration flow, resulting in the accumulation and redistribution of particulate plastics in plant tissues (Li et al., 2019). The uptake of particulate plastics has been observed in some plant species, as shown in **Table 2**. Bandmann et al. (2012) showed the uptake of nanoplastics into plant cells via a cell culture study. They found that BY-2 cells had taken up 20 nm nanobeads that were exposed for 15 min. However, the study was based on a plant cell culture experiment, which could not fully prove whether living whole plants would take up nanoplastics. A recent study by Li et al. (2019) found that PS beads with the size of 200 nm were transferred from the roots to stems and leaves of lettuce (*Lactuca sativa L., Rosa*) plants, which indicated that the whole plant could accumulate particulate plastics. The above results were supported by Sun et al. (2020) who observed that both negatively charged (PS-SO_3H, 55±7 nm) and positively charged (PS-NH_2, 71±6 nm) PS nanoplastics were taken up by *Arabidopsis thaliana*, providing direct evidence that nanoplastics could be absorbed and accumulated inside terrestrial plant bodies.

Early studies believed that micrometer and sub-micrometer plastic particles were difficult to be taken up by plants, and most of the particulate plastics would stay in the epidermal cells of the root system (Bosker et al., 2018). With the development of sophisticated particulate plastic analysis techniques, significant progress has been made recently to detect plant uptake of micro- and nanoplastics. Li et al. (2020b) found that plastic particles of submicron (0.2 μm) or even micron size (2.0 μm) could penetrate the roots of wheat (*Triticum aestivum*) and lettuce (*L. sativa*), and enter the edible parts of the crops through water and nutrient flow under the action of transpiration force. This result implies that particulate plastics can accumulate in plants in the real environment, suggesting that micro- and nanoplastics may exist in our daily diet through edible crops (Zhang et al., 2020b). Rillig (2020) stated that the uptake of particulate plastics by plant roots could only occur in hydroponic culture but not in soil culture experiment. Taylor et al. (2020) found that even in hydroponic culture, particulate plastics (40 nm~1 mm) could not be taken up by wheat roots (*T. aestivum*). Regardless of the fact that particulate plastics are uptaken by plants or not, protecting croplands from potential plastic pollution is crucial, as discussed later in this paper.

At present, beads are the predominant forms of particulate plastics used in plant studies by most researchers (**Table 2**), which might be due to the easy commercial availability of standard bead particles (Rillig et al., 2019). Various forms of particulate plastics could be present in agricultural soils, especially the plastic fibers make up to 92% of the particulate plastics (Zhang and Liu, 2018). Thus, plant uptake of other forms of particulate plastics, including fibers, need to be investigated in the future. Although it has been confirmed by spectroscopic techniques that particulate plastics could be distributed in plant tissues such as roots, stems and leaves, no relevant report is available on the quantitative microplastic concentrations and microplastic types taken up by plants. This fact could be due either to a lack of research, or a lack of effective and standardized methods for the separation and identification of particulate plastics in plant tissues. Therefore, it is necessary to improve analysis techniques of particulate plastics in plants in the near future, which would be the key to food safety concerning microplastic pollution.

3.2 Mechanisms of particulate plastic uptake by plants and redistribution

Although the uptake, accumulation, and redistribution mechanisms of particulate plastics in plants are unclear, it is likely that endocytosis (Bandmann et al., 2012), apoplastic transport (Sun et al., 2020) and crack-entry mode (Li et al., 2020b) are responsible in the above phenomena (Fig. 2).

3.2.1 Endocytosis

Bandmann et al. (2012) found that nano-beads (20 nm and 40 nm) were internalized rapidly by walled BY-2 cells via endocytosis, while large size particulate plastics (e.g., above 100 nm) were mostly excluded from internalization because the diameter of endocytic vesicles typically is in the range of 70 nm to 180 nm which is too small to internalize the large particles. Compared with wall cells, BY-2 protoplast cells could internalize larger nano-beads with a size up to 1000 nm because BY-2 protoplast cells could form larger endocytic vesicles. Bandmann et al. (2012) reported that clathrin-dependent endocytosis promoted the uptake of nano-beads in BY-2 protoplast cells. Unfortunately, the endocytosis mechanism described above was based on the design of independent cell culture, and no follow up study is available till date.

3.2.2 Apoplastic transport

Once particulate plastics enter the plant roots, some of the particles are captured by the mucus (highly hydrated polysaccharide) layer of roots, concentrating the particles on the root surface, and then transporting them in plant tissues through apoplastic transport (Sun et al., 2020). The dominant driving force for the apoplastic transport is the transpirational pull, which significantly promotes the allocation of particulate plastics in plant tissues (Li et al., 2019). The apoplastic transport from the cortex to the vascular bundle is impeded by the endodermic Kasparian strip, which obstructs the penetration of pollutants (Schreiber et al., 1999). Therefore, pollutants on the apoplastic route are forced to pass through the endodermic plasmalemma (Wang et al., 2020a). Li et al. (2019) found that 200 nm PS beads were mainly located in the vascular system and on the cell walls of the cortex tissues of lettuce (*L. sativa*) roots. Another study involving *A. thaliana* found that negatively charged nanoplastics (PS-SO_3H, 55± 7 nm) were internalized into the stele via apoplastic pathway, but such phenomenon was not observed in the case of positively charged nanoplastics (PS-NH_2, 71± 6 nm) (Sun et al., 2020). Unfortunately, the above results did not adequately confirm whether nanoplastics could be absorbed by plants via apoplastic transport, because the particles size (70 nm) used in the above study did not fully represent the size range of nanoplastics. Additionally, many plastic polymers with long alkyl chains and high molecular weight (i.e., high octanol-water partition coefficient (log K_{ow}) value and low water solubility) are difficult to be taken up by plants through apoplastic pathway (Gao and Collins, 2009). The above discussion shows that even if the particle size meets the requirements of plant uptake, the molecular structure of plastics may not meet the requirements of uptake. Therefore, plastic particle uptake by plants through apoplastic transport is closely related to the physical and chemical properties of the plastic particles, which needs future research to further understand.

3.2.3 Crack-entry mode

A breakthrough has been claimed recently by Li et al. (2020b) in understanding the mechanisms behind plant uptake of particulate plastics, where a physical access channel for particulate plastics to bypass the apoplastic pathway into wheat (*T. aestivum*) plants was observed. In most cases, since the diameters of cell wall pores and intercellular plasmodesmata are 3.5-5.0 nm and 50-60 nm, respectively (Smith, 1978; Carpita et al., 1979), nanoplastics larger than 5 nm would not penetrate the plant cell wall, and nanoplastics larger than 60 nm would not diffuse into the intercellular space. However, particulate plastics with large size (e.g., 200 nm) were reported to penetrate through the cell wall by the root cap mucilage which entrapped the particulate plastics in root cell wall (Li et al., 2020b). During active cell division, the apical meristem tissues were highly porous, and such physical characteristics enabled the diffusion of particulate plastics through the apical meristem tissues. Additionally, some cracks between the epidermal cells and sites of lateral roots could emerge during the cell separation, which would provide a transport crack for microplastics (e.g., 2.0 μm) to penetrate the stele (Li et al., 2020b). Once inside the stele, particulate plastics could transport towards the aboveground plant parts through the xylem along with the transpiration stream (Li et al., 2020b). It is worth noting that even though some large-size particles cannot pass through the cell wall pores and intercellular plasmodesmata, some intrinsic nature such as weak stiffness of plastic particles could lead to extrusion and deformation caused by intracellular internalization (Li et al., 2019). The mechanical flexibility of particulate plastics might be essential for their uptake via the crack-entry mode (Li et al., 2020a). However, more research is needed to further establish the crack entry mechanism of nanoplastic entry into plant bodies. In fact, a number of possible mechanisms could jointly affect the uptake of particulate plastics by plants, and more types of particulate plastics and plants need to be considered in future studies.

3.3 Factors affecting particulate plastics uptake by plants

3.3.1 Size of particulate plastics

The accumulation and translocation of particulate plastics in plants mainly depend on the particle size of the particulate plastics. For example, 20 and 40 nm nano PS beads were taken up by BY-2 cells, while 100 nm beads were excluded from uptake into turgescent and plasmolysis cells (Bandmann et al., 2012). Jiang et al. (2019) demonstrated that 100 nm PS fluorescent nanoplastics were accumulated in *Vicia faba* roots, while most of the particles blocked the cell wall pores. This phenomenon was consistent with the report of Li et al, (2019) that 1.0 μm PS beads were not taken up by lettuce (*L. sativa L., Rosa*). PS beads of 0.2 μm size mainly located in the vascular system and on the cell walls of the cortex tissues of the lettuce roots. The large particle size plastics were difficult to enter plant cells due to the permeability of the cell wall, and most of them were adhered on the surface of plant roots. The small size plastic particles, especially nanoplastics entered into the root cells easily, and passed through the intercellular space to translocate and accumulate elsewhere (Jiang et al., 2019). In fact, until now, due to the limitations of particulate plastics detection technology in plants, it is difficult to judge the actual particle size of particulate plastics which can be taken up by plants. For instance, Sun et al. (2020) studied particulate plastics of different sizes, and found that the PS nanoplastics with the size of less than 200 nm were taken up by *A. thaliana*, but the experiment did not demonstrate which precise particle size of particulate plastics were mainly taken up by the plant.

3.3.2 Type of particulate plastics

The types of particulate plastics are also a key factor affecting their accumulation and translocation in plant bodies. For example, Sun et al. (2020) reported that positively charged PS nanoplastics (PS-NH_2) were accumulated more than negatively charged nanoplastics (PS-SO_3H) by *A. thaliana*. Because of the charged characteristics of plant cell membranes, when micro- or nanoplastics with electrical charge cross the plant cells, they compete with other charged ions for the adsorption sites and thus may be excluded from the cell membranes (Miller et al., 2016). Therefore, the charge characteristics of micro- and nanoplastics are closely related to the particles' plant uptake. In addition, the physiological characteristics of plants, such as root exudates, are a key factor affecting the absorption of micro- and nanoplastics in plants (Li et al., 2020b). Because of the strong adhesion of plastic particles, particulate plastics are easily "captured" by the polysaccharide mucus excreted by plant roots (Li et al., 2020b). Moreover, the aggregation (Wang et al., 2021a) of particulate plastics is significantly increased by root exudates (Sun et al., 2020), which resulted in decreased mobility of particulate plastics in soil (Wang et al., 2021b), and ultimately prevent the particulate plastics uptake by plants.

4 Effects of particulate plastics on plants

Particulate plastics taken up by plants not only cause potential food safety problem, but also have a certain impact on plant traits. Although the effects of particulate plastics on aquatic organisms is substantially evident, there is no indisputable evidence of the plant impact of particulate plastics till date. Such effects are discussed in three parts: community-level effects, individual-level effects, and cell-level effects. Table 3 gives an overview of the particulate plastics' effects on plants concerning the microplastic type, particle size, concentration, influence location, and effect phenomenon.

4.1 Community-level effects

At community-level, the community evenness of plants could be affected by particulate plastics, and plant synergetic interactions could become out of balance, which might result in few species to dominate the ecosystem function (Poeta et al., 2017; Rillig et al., 2019). Such increase of community evenness could change plant diversity and community composition, and lead to decreased ecosystem functionality. Lozano and Rillig (2020) found that due to the reduction of soil bulk density and increasing soil macroporosity by microfibers exposed in soil, the shoot and root mass of grasses and herbs increased, which led to the invasion of *Calamagrostis* in Europe, and the allelophatic *Heieracium* became a dominant species. The alteration of interaction between invasive plant species and local species caused by particulate plastics might lead to changes in soil associated bacterial and fungal species, and such variations tend to accelerate the carbon cycle. This might lead to excessive carbon loss and a serious ecological risk (Waller et al., 2020). However, due to limited short-term experimental observations, there is insufficient evidence for the conclusion of the impact of particulate plastics at plant community level (Lozano and Rillig, 2020). Studying the long-term responses of particulate plastics on plant community is a need of the hour.

4.2 Individual-level effects

Compared with the community-level effects, the impact of particulate plastics on plants is more focused at the individual level. The effects of particulate plastics on physiological and biochemical characteristics of plants at individual level are shown by apparent characteristics such as seed germination and plant growth parameters. Bosker et al. (2019) studied the effects of particulate plastics on seed germination, and found that the germination rate of cress (*Lepidium sativum*) seeds was significantly inhibited by plastic particles with the size of 50,500, and 4,800 nm, and the negative effects increased with the increased concentration of particulate plastics exposed (Bosker et al., 2019). The reason for the decrease of germination rates could be via blocking the inner capsule of seeds with particulate plastics. Such results imply that the short-term and transient negative influence of particulate plastics on terrestrial plants are not enough to fully reveal the community-level effects.

A large number of higher plants, such as lettuce (*L. sativa L. var. ramosa Hort*), broad bean (*V. faba*), wheat (*T. aestivum*), spring onion (*Allium fistulosum*), maize (*Zea mays L. var. Wannuoyihao*), and rice (*Oryza sativa* L.), have shown a certain influence caused by particulate plastics (e.g., de Souza Machado et al., 2019; Gao et al., 2019; Jiang et al., 2019; Li et al., 2020a; Wang et al., 2020b; Wu et al., 2020). Studies have demonstrated the impact of particulate plastics on traits of plant roots and leaves. For example, 2% (w/w) polyethylene high density (PEHD), PET, PS, PES, PA and PP particulate plastics increased the root length and root area of spring onion *(A. fistulosum)*, while decreased the root average diameter (de Souza Machado et al., 2019). However, the results of root biomass response were different. Boots et al. (2019) and de Souza Machado et al. (2019) found that root biomass was significantly increased by particulate plastics, whereas Qi et al. (2018) found an opposite effect where the root biomass of wheat (*T. aestivum*) was significantly decreased by low-density polyethylene (LDPE, 1%) and starch-based biodegradable plastics (1%).

Stems and leaves play an important role in the long-distance transportation of water and nutrients for plant growth, so obvious responses in stem and leaf structures or components might have consequences on the plant growth (Gao et al., 2009). Compared to the root system, the effect of particulate plastics on plant leaves was less significant. The effects on leaves mainly manifested as inhibition of growth, hindrance of the chlorophyll fluorescence and interference with the antioxidant defense system (Gao et al., 2019; Li et al., 2020c), thereby impacting one of the most important plant physiological functions, i.e., photosynthesis. The water content and C / N ratio (de Souza Machado et al., 2019), chlorophyll content (Qi et al., 2018; Boots et al., 2019; Wang et al., 2020b), and enzyme activity (Jiang et al., 2019) of plants were significantly altered under microplastic stress, which in turn might influence the plant growth. The variety of individual plant responses indicates that the environmental behavior of particulate plastics in the soil ecosystem is complex, and the apparent and visible physiological responses might be a manifestation of stress at the cellular and molecular levels.

4.3 Cell-level effects

Early studies on algae showed that particulate plastics could induce cell wall damage and cell maturation cracking (Zhang et al., 2017; Mao et al., 2018). Cell damage, interference with the intracellular molecules, and oxidative stress caused by particulate plastics were also found in the cells of higher plants (Gao et al., 2019; Jiang et al., 2019; Rillig et al., 2019). Superposition of particulate plastics may block the root cell, leading to toxic effects (Gao et al., 2019; Jiang et al., 2019). Jiang et al. (2019) demonstrated that cell wall pores of *V. faba* were blocked by PS particles of 100 nm size, which led to a decrease in the enzymatic activities. Additionally, Zhang et al. (2019) demonstrated that hydroxybenzoic acid was significantly decreased by PS, which led to the alteration of cell wall compositions in plant (*Spinacia oleraceae*). Reactive oxygen species (ROS) are important indexes in the study of cytotoxic effect, which can give rise to damage of the cell structure and functions (Zhang et al., 2011). Nanoparticles are often found to facilitate production of ROS that can cause oxidative stress on higher plants and algal cells (Jiang et al., 2019). The stress of ROS can affect the energy metabolism of plants by reducing the degree of anabolism (Alscher et al., 1997; Wu et al., 2020). Such phenomenon was confirmed by Sun et al. (2020) who found that particulate plastics downregulated the metabolic processes of ROS. The stress of ROS might also affect the gene toxicity of plants (Rice-Evans et al., 1997; Paré and Tumlinson, 1999; Jiang et al., 2019). For example, a micronucleus (MN) test showed that PS (5 μm and 100 nm) increased MN frequency in cells of *V. faba* roots, indicating increased genotoxic potential caused by particulate plastics (Jiang et al., 2019). The upregulated gene ontology (GO) including biosynthetic and metabolic processes could eliminate radicals in *A. thaliana* under the PS-NH_2 stress, making the plants more adaptable to environmental changes and relatively less affected by particulate plastics (Sun et al., 2020). This conclusion was drawn based on the finding that the root epidermal cells of *A. thaliana* were altered by nanoplastics and resulted in swelling of the root maturation zone. The above led to a downregulation of water deprivation related gene, and hence decreased the fresh weight of *A. thaliana*. However, sufficient evidence is not available to reveal the mechanism of plant molecules regulating the stress of particulate plastics. Therefore, to reveal the molecular regulatory mechanisms of plants under the stress of particulate plastics, more extensive research on the interaction between particulate plastics and plants need to be carried out.

5 Factors of particulate plastics' effect on plants

5.1 Plant species

Different plant species show different effects with the same type of particulate plastic treatment. PES fibers (30 µm, 0.4%) increased the shoot mass of *Calamagrostis*, while decreased the shoot mass of *Holcus* (Lozano and Rillig, 2020), which could be attributed to the difference of response mechanism for plants under the pollution stress (Zhao et al., 2018; van Weert et al., 2019). Under the coercion of exogenous substances, plants adapt to external pressure by adjusting their physiological and biochemical structures. Once the stress caused by exogenous substances exceeds the tolerance range of plants, it will have a serious toxic effect on plants (Gao et al., 2019). Therefore, the traits of plants would be changed under the exposure to particulate plastics. A wide range of plant traits (e.g., root biomass, root length) exist among different plant species. For example, van Weert et al. (2019) found that *M. spicatum* had extensive root system in the sediments compared to *Elodea sp.*, which enhanced the potential to interact with particulate plastics, and thus hindered the translation of nutrients by roots, and ultimately reduced root elongation. Moreover, some plant may be more susceptible to exogenous substances (Rillig, 2020). Therefore, attention should be paid to the species-specific effects while examining the effects of particulate plastics on plants by determining the differences in plant traits.

5.2 Plastic types

The characteristics of particulate plastics such as their shape, size, type and dose could also significantly influence the effects of particulate plastics on plants (Lozano et al., 2020; Van Weert et al., 2019). Two shape-related mechanistic effects of particulate plastics on plants, namely shape dissimilarity and shape mediation effects, were proposed by Lozano et al. (2020). The shape dissimilarity mechanism proposes that greater the diversity between shape form of particulate plastics and soil medium, the greater the impact on the soil systems (such as soil structure, properties plant and microorganism). Due to the linear structures, fibers are substantially different to the non-linear particles that composed the bulk of soil mass, and hence fibers might have higher potential to effect on soil biophysical properties (de Souza Machado et al., 2018). The shapes of PES fibers were more dissimilar to the natural particles of a tested sandy loam soil than that of PEHD particles, hence the soil structure was more strongly affected by PES fibers than PEHD particles, causing significant responses to spring onion (*A. fistulosum*) by the former type of particulate plastics (de Souza Machado et al., 2019). The shape mediation mechanism suggested that apart from particle shape, surface properties, composition and additives of particulate plastics were the key influencing factors (Lozano et al., 2020). Particulate plastics with same shape but with different properties would also vary in their effects on plant traits (Lozano et al., 2020). Kalčíková et al. (2017b) found that although a significant reduction in duckweed (*Lemna minor*) population was caused by both sharp- and soft-edged PE microbeads, the former particles caused more root damages than the latter particles.

5.3 Plastic particle size

In general, the smaller the particle size of nanoplastics, more easily they are taken up by plants (Rillig et al., 2019). The small size plastic particles might cause a series of stress reactions leading to significant plant responses (Jiang et al., 2019; Li et al., 2019; Li et al., 2020c; Sun et al., 2020). Li et al. (2020c) found that the effects on root morphology, antioxidant system, and photosynthesis fluorescence parameters of lettuce exposed to PVC particles of 100 nm to 18 μm size range were more significant than those exposed to PVC with particle size ranging from 18 μm to 150 μm. This was likely due to the poor mobility of particulate plastics with large particle size. The particulate plastics with small particle size were more likely to be captured by root epidermal polysaccharides resulting in physical and/or chemical toxic effects (Jiang et al., 2019; Li et al., 2020b). For example, higher genotoxic and oxidative damage were produced by 100 nm PS fluorescent nanoplastics compared with 5 mm PS fluorescent microplastics (Jiang et al., 2019).

5.4 Exposed concentration of particulate plastics

The concentration of particulate plastics in soil is another important factor affecting plant growth. Low concentration of particulate plastics, which could be diluted and dispersed by the soil to be absorbed by plant roots, might bring a positive effect on root traits, while high concentration might cause toxic effects on crops and inhibit plant growth (Li et al., 2020c). Interesting results were reported by Wang et al. (2020b) where no noticeable phytotoxicity of PE particulate plastics on maize (*Z. mays*) was observed even at high particle concentration, but polylactic acid (PLA) particles exhibited a high phytotoxicity at 10% dosage. It was concluded that phytotoxicity of particulate plastics was shown when the concentration of the particles was increased to a certain extent. This conclusion was confirmed by Jiang et al. (2019) who found no inhibitory influence of fresh and dry weight of *V. faba* exposed to 10 and 50 mg/L PS nanoplastics (100 nm), while a significant inhibitory effect was observed at 100 mg/L concentration. However, the concentration of particulate plastics in the real soil environment is not as high as the above experimental design. The highest weight of particulate plastics found in soil has been 7% reported so far (Fuller and Gautam, 2016). Therefore, although an intimate link between microplastic concentration and response to plants were observed, the relationship of dosage effect is not yet clear. In summary, the responses of plants are closely related to the characteristics of particulate plastics (i.e., type, size, shape and dose) and plants (i.e., species) (Gao et al.,2019).

6. Mechanisms of particulate plastics' influence on plants

The influence of particulate plastics on plant performance occurring in the complex soil-plant system is underpinned by soil physical and chemical properties, regulation of rhizosphere microenvironment, plant nutrient transport system, and direct toxicity (Rillig et al., 2019).

6.1 Changing soil physicochemical properties

The main mechanism of the effect of particulate plastics on plants may be through regulating the physicochemical properties of soil, thus indirectly affecting the environment of plant growth (Chen et al., 2020; Kumar et al., 2020). Particulate plastics can change soil biophysical environment, i.e., soil bulk density, soil aggregation, and water dynamics, resulting in the adjustment of plant traits to new soil conditions (de Souza Machado et al., 2019; Sanchez-Hernandez et al., 2020; Kumar et al., 2020). For example, higher water stable aggregates and soil bulk density were observed in the rhizosphere of spring onion (*A. fistulosum*) with the presence of PES, PET and PP, while the bulk density of the soil was decreased (de Souza Machado et al., 2019). Soil bulk density reduction and soil macroporosity increase due to microfiber addition in soil promoted roots to better penetrate into the soil matrix, which ultimately increased the root biomass (Lozano and Rillig, 2020). Such ameliorations in soil may promote root penetration and ultimately stimulate root growth. Sun et al. (2020) reported that PLA might have more profound impacts than PE on soil properties, particularly on soil pH, and thus could alter nutrient availability. This might further interfere with the transport of essential nutrients for chlorophyll synthesis in leaves, indicating that the soil biophysical environments are strongly affected by particulate plastics. Moreover, particulate plastics could be weathered by light, wind, water and microorganisms, and the effects of particulate plastics on plants might change over time (Yoshihisa et al., 2012; Gao et al., 2019).

6.2 Changing rhizosphere environment

The changes of rhizosphere microenvironment caused by particulate plastics exposure, such as root exudates and microbial community structure, directly affect the growth environment of plants. For example, particulate plastics can affect the production of secondary metabolites such as volatile compounds in the rhizosphere, significantly affect the dodecanal emission in the rhizosphere (Qi et al., 2020), which is believed to be harmful to both fungal and plant growth. Volatile emission in the rhizosphere can induce or inhibit plant growth through a modulation of the metabolism, hormonal balance and nutrient intake of plants (Fincheira and Quiroz, 2018). Additionally, particulate plastics could affect the secretion of root exudates, e.g., oxalic acid, which promote the aggregation of particulate plastics, and finally affect their mobility and uptake (Sun et al., 2020). The effect of particulate plastics on microbial community structure and abundance in soil is another important aspect of plant growth (Lozano et al., 2020). Due to the increasing mortality and histological damage of soil microorganisms, and reduction of richness and diversity of bacterial communities, the net soil microbial activity is significantly affected by particulate plastics. The richness of *Gematimonas* (one of the essential genera of phosphate solubilizing bacteria) was increased by membranous PE and fibrous PP (Yi et al., 2020), which promoted the dissolution of unavailable phosphorus leading to the increase of plant available phosphorus. PE and PVC stimulated the bacteria associated with nitrogen fixation (Fei et al., 2019) directly affecting the utilization of nitrogen by plants (Chen et al., 2015). Therefore, the nitrogen cycle and phosphorus cycle in soils were impacted to some extent by particulate plastics (Yi et al., 2020), which could lead to altered soil quality and variation in plant responses. Additionally, proteobacteria abundance in soil was promoted by particulate plastics (Huang et al., 2019), which might potentially promote plant growth (Fierer et al., 2007; Hortal et al., 2013).

Another mechanism of the effect of particulate plastics on plants is to regulate the activity of arbuscular mycorrhizal fungi (AFM) in the rhizosphere. AMF obtains essential carbohydrate and other nutrients from plant roots, simultaneously promotes the absorption of water and nutrients by plants, forming a synergistic relationship with plants (Bolan, 1991; Berruti et al., 2016). In fact, the abundance of AMF hyphae was significantly increased by PES, which stimulated the growth of spring onion (*A. fistulosum*) (de Souza Machado et al., 2019). To sum up, understanding the intervention of particulate plastics in the rhizosphere environment is important to unravel the mechanism of particulate plastics' influence on plants. Although the effects of particulate plastics on soil or plant system have been separately acknowledged in some studies, the inter-relationships among the effects of particulate plastics on soil and plants together are still lacking.

6.3 Nutrient (im)mobilization

Particulate plastics could interfere with the absorption of nutrients and water by plants (Van Weert et al., 2018; Rillig et al., 2019). Some particulate plastics carry abundant C (e.g., PS, PE) (de Souza Machado et al., 2018; Rillig, 2018), which could indirectly change the C allocation of plants belowground (Rillig, 2018； Zang et al., 2020; Zhou et al., 2021). The change in plant C allocation caused by particulate plastics might alter the symbiosis of microbial communities and plant mycorrhiza, and affect the activities of C, N, P-related enzymes (Zhou et al., 2021), ultimately affecting the plant growth. Although the contents of N and P compared with C are negligible in particulate plastics (de Souza Machado et al., 2018), they could also alter the transformation of nutrients mediated by soil microorganisms (Zhou et al., 2021). Moreover, a high C:N ratio is observed in the soil due to microplastic contamination, inducing an increase in microbial immobilization of nutrient elements (Rillig et al., 2019).

On the other hand, the root system provides enormous aggregation sites for particulate plastics, which results in t blockage of the root surface pores, eventually hindering the absorption of essential nutrients by plants (Gao et al., 2019; Jiang et al., 2019). In a certain range of particle size, physical destruction of cells might occur due to the attachment of bigger size of particulate plastics to the cell surface of plants, leading to serious barrier to transport of nutrients and water (Bosker et al., 2019). The chemical/ toxic effects on plants are likely to increase when the particles size of particulate plastics is small which could increase the dissolution of particles (Li et al., 2020c).

6.4 Direct toxicity

The potential damage of cells, molecules and oxidative stress caused by particulate plastics might be due to the active regulation of plant adaptation to the new environmental stress (Rillig et al., 2019). Particulate plastics can adhere onto root surface, create blockage in the root cell space, and enter into plant tissues (Gao et al., 2019; Sun et al., 2020; Wang et al., 2020b), which result in community-level effects, individual-level effects and cell- level effects. Therefore, the mechanism of influence of particulate plastics on plants may be caused by direct toxicity too (Rillig et al., 2019). The direct toxic effects of particulate plastics on plants are reflected in two aspects: physical damage and biochemical toxicity (Li et al., 2019; Li et al., 2020c; Wu et al., 2020). Due to the disruption of nutrient absorption by roots caused by the altered cell wall components, the activity of antioxidants in rice leaves was decreased by particulate plastics (Wu et al., 2020). This suggests that because of the damage of cell structure, nutrient accumulation could probably be hindered by the exposure of nanoplastics (Van Weert et al., 2019). Such physical damage might destroy the integrity and functionality of the cells, leading to the production of a variety of responses in plants.

The most obvious aspect of the biochemical toxic effect of particulate plastics on plants is the redox system (Li et al., 2020c). Particulate plastics could increase the levels of ROS in plants

(Gao et al., 2019). ROS can affect the metabolic pathways including energy metabolism and anabolism (Alscher et al., 1997). To deal with the cytotoxic effects, plants initiate a series of antioxidant reactions that increase the enzymatic activities to eliminate ROS (Alscher et al., 1997; Wang et al., 2010). Wu et al. (2020) studied the potential effect mechanism of PS on rice from the perspective of metabolic system, and showed that particulate plastics reduced the biosynthesis of amino acid, nucleic acid, fats and some secondary metabolites by excessive formation of ROS beyond the scavenging capacity of the antioxidant system, and thus leading to decreased membrane activity. Therefore, to adapt to the stress of ROS, plants would regulate the enzymatic activities to avoid possible oxidative damage (Paré and Tumlinson, 1999). Wu et al. (2020) found that the accumulation of ROS in rice (*O. sativa* L.) leaves after exposure to PS was very small, which was consistent with lower activities of superoxide dismutase (SOD), peroxidase (POD), and catalase (CAT) activities than those of the antioxidant enzymes.

6.5 Bioturbation and decomposition by earthworms

Earthworms are widely distributed in the terrestrial ecosystem, and play an important role in maintaining soil health, such as improving the soil structure and soil permeability, drainage and water holding capacity, and thus change the plant growth environment (Edwards, 2004). Additionally, earthworms can promote the activity of soil microorganisms (Dempsey et al., 2013; Hoang et al., 2016). Therefore, the soil physicochemical and biological features which might be changed by earthworms directly affect the growth of plants. Another important influence of earthworms in soil is their interference with the fate of particulate plastics. Particulate plastics mixed in litter are ingested/egested by earthworms and then carried to the subsoil, such bioturbation would redistribute the residence of particulate plastics in soil (Huerta et al., 2016;

Rillig et al., 2017). Thus, the contact of particulate plastics with roots in rhizosphere may be strongly impacted. Moreover, earthworms can induce decomposition of biodegradable plastics (Sanchez-Hernandez et al., 2020), which lead to the decrease of the particle size of biodegradable plastics, and may eventually make them reach the size that can be absorbed by plants. Therefore, from the perspective of food safety, earthworms might increase the risk of plants to accumulate particulate plastics. From the perspective of agricultural production, earthworms might improve the plant growth by decomposing plastics and promoting the soil nutrient cycling. For example, plant growth indices, including the biomass of root and shoot, leaf area, number of leaves and relative chlorophyll content, of wheat (*T. aestivum*) were significantly increased by earthworms when exposed to particulate plastics because earthworms alleviated the nutritional impairment in wheat plants caused by particulate plastics (Qi et al., 2018).

7 Interaction of particulate plastics and environmental pollutants

The interaction of particulate plastics with environmental pollutants (i.e., heavy metal(loid)s and organic chemicals) include two aspects: (1) adsorption/desorption of heavy metal(loid)s and organic pollutants by particulate plastics (Akhbarizadeh et al., 2017; Bradney et al., 2019), and (2) release of chemically active substances inherently contained in particulate plastics (Bradney et al., 2019; Zhou et al., 2019b; Menéndez-Pedriza and Jaumot, 2020). Some environmental pollutants, i.e., antibiotics, polycyclic aromatic hydrocarbons, and Cd were locally enriched on microplastic surfaces (Hüffer and Hofmann, 2016; Li et al., 2018b; Godoy et al., 2019; Patterson et al., 2020). The above two sources would eventually undergo desorption and release pollutants, leading to a combined pollution of environmental contaminants and particulate plastics in the environment (Hartmann et al., 2017; Menéndez-Pedriza and Jaumot, 2020).

Via adsorbing a considerable quantity of heavy metal(loid)s and organic pollutants, particulate plastics play an important vector role for pollutant transport in the environment, producing a significant impact on the biogeochemical cycling of pollutants, especially in aquatic ecosystems (Hartmann et al., 2017; Liu et al., 2018b; Bradney et al., 2019). Akhbarizadeh et al. (2017) found that the concentrations of Cd, Mn, Zn, As and Pb in coastal sediments were positively correlated with the concentrations of particulate plastics. The combination of particulate plastics and environmental pollutants could cause changes in the histology, molecular functions, cytology and behavior of animals, as have been widely researched in recent years (Hartmann et al., 2017).

At present, there are only few studies on the effects of combined pollution of particulate plastics and environmental pollutants on the soil ecosystem. Early studies found that high-density polyethylene (HDPE) increased the bioavailability of Zn in earthworms, increasing the possible hazard to the soil ecosystem (Hodson et al., 2017). Zhou et al. (2020) observed a similar phenomenon, where microplastic particles increased the accumulation of Cd in earthworms. These results indicated that particulate plastics could increase the risk of heavy metal(loid)s to soil animals. The bioaccumulation of some chemicals in mixture (e.g., additives, plasticizers) was increased through particulate plastics, while other chemicals' (i.e., persistent organic pollutants (POPs)) bioaccumulation was decreased by particulate plastics in aquatic organisms at the same time (Koelmans et al., 2015). Therefore, further studies are required to ascertain the migration and transformation of environmental pollutants via particulate plastics in soils.

Another important aspect of the interaction between particulate plastics and environmental pollutants in soils is reflected through the plant uptake of pollutants from soils (Abbasi et al., 2020; Gao et al., 2020). The interaction of particulate plastics with other environmental pollutants in soils might affect the bioavailability of the pollutants, leading to a change in the uptake of pollutants in plant bodies (Abbasi et al., 2020; Kumar et al., 2020). For example, PET particles promoted the transfer of heavy metals (Zn, Cd, and Pb) to the wheat rhizosphere zone (Abbasi et al., 2020), which subsequently facilitated a direct contact between plant roots and the metal ions, increasing plant uptake. Dong et al. (2020) demonstrated that the uptake of As (III) in rice was affected by particulate plastics through three distinct ways, including a direct adsorption of As (III) ions by particulate plastics, competition between particulate plastics and As (III) ions for adsorption sites on the root surface, and inhibition of root activity.

However, some contrary findings were found in some studies, where bioavailability of pollutants in soil was increased by plastics, but the contents of pollutants in plants did not increase as expected (Dong et al., 2020; Wang et al., 2020b). The bioavailability of Cd in soil was increased by particulate plastics, but no increase in Cd content was found in maize (Kirkham, 2020; Wang et al., 2020b). Gao et al. (2020) found that when exposed to 0.25 to 1.00 mg/mL of PE, the contents of di-n-butyl phthalate (DBP) in green lettuce leaves and roots were decreased respectively by 11.24-30.80% and 13.26-30.80% compared with sole DBP exposure. In purple lettuce, the respective decreases of DBP concentrations in leaves and roots were 7.18-23.37% and 9.39-29.72%. Particulate plastics may physically obstruct the contact of pollutant compounds with roots due to the attachment of particulate plastics onto the root surfaces, and interfere with the uptake of pollutants from soil solution, likely because of the hydrophobic property of particulate plastics (de Souza Machado et al., 2019; Wang et al., 2020b).

In the case of aquatic organisms, it is widely accepted that a combined pollution of particulate plastics and other pollutants might bring more adverse ecological risks than particulate plastics alone (Hartmann et al., 2017; Wang et al., 2020b; Sridharan et al., 2021). Zhu et al. (2019) found that coexistence of particulate plastics (PE, PS, and PVC) and triclosan induced oxidative stress, and led to inhibition of microalgae growth. Dong et al. (2020) found the combination of particulate plastics and As (III) increased O_2^- and H_2O_2 in rice roots and leaves, which led to the induction of lipid peroxidation and damage of cell membranes. Similarly, photosynthesis was inhibited by coexistence of PS and As (III), which in turn significantly decreased rice biomass (Dong et al., 2020). Moreover, a large number of additives added in the plastic production process would be gradually released into the environment following the weathering of plastics (Staniszewska et al., 2016; Duan et al., 2021), which might cause secondary pollution of the environment. The migration and transformation of additive compounds could be altered by particulate plastics, which might lead to alarmingly toxic effects of particulate plastics and their complexes on the ecosystem (Dong et al., 2018).

Particulate plastics thus could increase other environmental pollutants' toxicity to plants. However, there is little research on the interaction between particulate plastics and other pollutants in the terrestrial ecosystem, especially on plants, which warrants immediate research attention globally.

8 Influence of biodegradable plastics on plants

With the development of research on particulate plastics in soil, environmentally friendly biodegradable plastics have emerged rapidly as agricultural mulch films to reduce the residues of plastics in soils. However, there are many problems caused by degradable plastic films. A complete degradation of plastic films needs specific environmental conditions and very long time (Rillig et al., 2019). Therefore, before completely biodegraded, plastics are mechanically disintegrated into nanoparticles during the weathering process (Pleiter et al., 2019), which increases the risk of plant uptake (Jiang et al., 2019). During the biodegradation process, soil properties, particularly pH, are strongly affected, which might reduce the bioavailability of nutrients and subsequent plant uptake of essential elements (Wang et al., 2020b). For example, the degradation of DF04P films (i.e., degradable plastic produced from corn starch) increased the pH value of soil from 7.91 to 8.29, which adversely effected the plant growth environment (Bettas et al., 2014).

Compared with non-degradable particulate plastics, degradable particulate plastics could have more obvious phytotoxicity to plants, which might be attributed mainly to the higher potential for the formation of nanoplastics, and the release of toxic additives and plastic monomers during the degradation process (Wang et al., 2020b). Qi et al. (2018) reported a stronger negative effect of biodegradable PLA particulate plastics on *T. aestivum* traits (e.g., grain biomass) than LDPE. Onion (*Allium cepa)* plants had shown molecular biological stress response (such as cytotoxic stress) when exposed to PLA contamination (Souza et al., 2013). For example, PLA significantly reduced leaf areas of *T. aestivum*, while LDPE had no significant effect on leaf area (Qi et al., 2018). The effect of biodegradable plastics on plant traits might be due to the potential stress of their degradation byproducts (e.g., lactic acid) (Boots et al., 2019).

PLA is a commonly used degradable plastic film, and can be degraded by microbes into lactic acid, which is a kind of root exudate and may participate in the secondary metabolism of plants, affecting the plant growth (Martin-Closas et al., 2014). For example, with 50 and 500 mg/L lactic acid, the shoot and root biomass of tomato (*Lycopersicon esculentum)* was significantly reduced (Martin-Closas et al., 2014). Moreover, PLA possibly incorporated in the soil, and affected the plant responses via nutrient immobilization by the degradation byproducts (e.g., lactic acid oligomers), which caused potential stress inhibiting the shoot length of *Lolium perenne* (Boots et al., 2019). Additionally, a large number of distinct volatile organic compounds such as dodecanal could be induced by PLA in the rhizosphere soil (Qi et al., 2020). Dodecanal has a negative effect on fungi (Wang et al., 2020b). Therefore, the activity and community structure of AMF were affected by dodecanal (Wang et al., 2020b). Moreover, the monomers and oligomers contained in degradable particulate plastics would be gradually released into the environment (Kim et al., 2003), and would subsequently alter the microbial community structure in the rhizosphere soil, affecting plant growth (Agarwal, 2020). In summary, biodegradable particulate plastics affect the variation of plant performances by changing the symbiotic associations in the plant-soil system (Wang et al., 2020b). The scope of applying biodegradable plastics as an alternative of mulch films in future agricultural practices needs to be carefully assessed.

9 Conclusions and future research priorities

This review shows that compared to the aquatic system, investigations of particulate plastics in the terrestrial ecosystem are deficient. Reported studies suggest that plants may uptake, accumulate and transport particulate plastics through crack-entry mode, endocytosis and apoplastic transport. Moreover, particulate plastics can cause significant responses in plants at the individual, cellular and molecular levels mainly due to the changes of soil physicochemical properties and rhizosphere environment, nutrient (im)mobilization and direct toxicity to plants following particulate plastics inclusion in the soil. However, current knowledge on the distribution, accumulation and transportation of particulate plastics in the soil-plant system is limited.

Based on the above observations, we propose several topics of research that need to be prioritized to understand the environmental behaviors of particulate plastics in farmland soils to ensure food security and food quality.

First, we need to establish efficient and rapid quantitative and qualitative methods for particulate plastic analysis in the soil and plants. Second, on the basis of convincing analysis methods, we need to understand the distribution characteristics and pollution levels of particulate plastics in the soil-plant system. We especially need to determine the real pollution level in plants under combined pollution of particulate plastics and other environmental contaminants, and link that information to plant responses. Third, we need to research on plant uptake of particulate plastics in the future, and reveal the mechanisms of plastic uptake by plants, such as the uptake pathways and uptake kinetics across different plant species and different types of particulate plastics. Fourth, we need to better understand the interaction of particulate plastics with environmental pollutants in the soil, and the role of such interaction in modulating the bioavailability of environmental pollutants to plants. Fifth, we need to investigate in detail the influence of particulate plastics on plants, in particular, the influence of type, size, shape, and content of particulate plastics. Finally, as for the degradable particulate plastics used in agricultural practices, we need future research to reveal the possible secondary environmental problems caused by the degradation byproducts, and develop best management practices for using and disposing agricultural plastic films.

Conflict of interests

The authors declare no intellectual or financial conflict of interests.

Acknowledgments

This study was financially supported by the National Natural Science Foundation of China (41907344; 21876027).

References

Abbasi, S., Moore, F., Keshavarzi, B., Hopke, P.K., Karimi, J., 2020. PET-microplastics as a vector for heavy metals in a simulated plant rhizosphere zone. Sci. Total Environ. 744, 140984. https://doi.org/10.1016/j.scitotenv.2020.140984.

Agarwal, S., 2020. Biodegradable polymers: present opportunities and challenges in providing a microplastic-free environment. Macromol. Chem. Phys. 221(6), 2000017. https://doi.org/10.1002/macp.202000017.

Akhbarizadeh, R., Moore, F., Keshavarzi, B., Moeinpour, A., 2017. Microplastics and potentially toxic elements in coastal sediments of Iran's main oil terminal (Khark Island). Environ. pollut. 220, 720-731. https://doi.org/10.1016/j.envpol.2016.10.038.

Allen, S., Allen, D., Phoenix, V.R., Roux, G.L., Durántez Jiménez, P., Simonneau, A., Binet, S., Galop, D., 2019. Atmospheric transport and deposition of microplastics in a remote mountain catchment. Nat. Geosci. 12(679). https://doi.org/10.1038/s41561-019-0335-5.

Alscher, R.G., Donahue, J.L., Cramer, C.L., 1997. Reactive oxygen species and antioxidants: relationships in green cells. Physiol. Plant. 100, 224e233. https://doi.org/ 10.1111/j.1399-3054.1997.tb04778.x.

Bandmann. V., Müller, J. D., Köhler, T., Homann, U., 2012. Uptake of fluorescent nano beads into BY2-cells involves clathrin-dependent and clathrin-independent endocytosis- ScienceDirect. FEBS Lett. 586(20), 3626-3632. https://doi.org/10.1016/j.febslet.2012.08.008.

Beriot, N., Peek, J., Zornoza, R., Geissen, V., Lwanga, E.H., 2021. Low density-microplastics detected in sheep faeces and soil: A case study from the intensive vegetable farming in Southeast Spain. Sci. Total Environ. 755, 142653. https://doi.org/10.1016/j.scitotenv.2020.142653.

Berruti, A., Lumini, E., Balestrini, R., Bianciotto, V., 2016. Arbuscular mycorrhizal fungi as natural biofertilizers: Let's benefit from past successes. Front. Microbiol. 6(426), 1559. https://doi.org/10.3389/fmicb.2015.01559.

Bettas Ardisson, G., Tosin, M., Barbale, M., Degli-Innocenti, F., 2014. Biodegradation of plastics in soil and effects on nitrification activity. A laboratory approach. Front. Microbiol. 5(2-3), 710. https://doi.org/10.3389/fmicb.2014.00710.

Bolan, N.S., Kirkham, M.B., Halsband, C., Nugegoda, D., Ok, Y.S., 2020. Particulate plastics in terrestrial and aquatic environments. CRC Press. https://doi.org/10.1201/9781003053071.

Boots, B., Russell, C.W., Green, D.S., 2019. Effects of microplastics insoil ecosystems: above and below ground. Environ. Sci. Technol. 53, 11496-11506. https://doi.org/10.1021/acs.est.9b03304.

Borrelle, S.B., Ringma, J., Law, K.L., Monnahan, C.C., Lebreton, L., Mcgivern, A., Murphy, E., Jambeck, J., Leonard, G.H., Hilleary, M.A., Eriksen, M., Possingham, H.P., Frond, H.D., Gerber, L.R., Polidoro, B., Tahir, A., Bernard, M., Mallos, N., Barnes, M., Rochman, C.M., 2020. Predicted growth in plastic waste exceeds efforts to mitigate plastic pollution. Sci. 369(6510). https://doi.org/10.1126/science. aba3656.

Bosker, T., Bouwman, L.J., Brun, N.R., Behrens, P., Vijver, M.G., 2019. Microplastics accumulate on pores in seed capsule and delay germination and root growth of the terrestrial vascular plant Lepidium sativum. Chemosphere 226, 774-781. https://doi.org/10.1016/j.chemosphere. 2019.03.163.

Bradney, L., Wijesekara, H., Palansooriya, K. N., Obadamudalige, N., Bolan, N. S., Ok, Y. S., Kirkham, M. B., 2019. Particulate plastics as a vector for toxic trace-element uptake by aquatic and terrestrial organisms and human health risk. Environ. Int. 131, 104937. https://doi.org/10.1016/j.envint.2019.104937.

Carpita, N., Sabularse, D., Montezinos, D., Delmer, D.P., 1979. Determination of the pore size of cell walls of living plant cells. Sci. 205, 1144-1147. https://doi.org/10.1126/science.205.4411.1144.

Chen, J., Zhou, H.C., Wang, C., Zhu, C.Q., Tam, N.F.Y., 2015. Short-term enhancement effect of nitrogen addition on microbial degradation and plant uptake of polybrominated diphenyl ethers (PBDEs) in contaminated mangrove soil. J. Hazard. Mater. 300, 84-92. https://doi.org/10.1016/j.jhazmat.2015.06.053.

Chen, C., Park, T., Wang, X., Piao, S., Xu, B.D., Chaturvedi, R.K., Fuchs, R., Brovkin, V., Ciais, P., Fensholt, R., 2019a. China and India lead in greening of the world through land-use management. Nat. Sustain. 2, 122-129. https://doi.org/10.1038/s41893-019-0220-7.

Chen, Y., Leng, Y., Liu, X., Wang, J., 2019b. Microplastic pollution in vegetable farmlands of suburb Wuhan, central China. Environ. Pollut. 257, 113449. https://doi.org/10.1016/j.envpol.2019.113449.

Chen, H.B., Yang, X., Wang, H.L., Sarkar, B., Shaheen, S.M., Gielen, G., Bolan, N., Guo, J., Che, L., Sun, H.L., Rinklebe, J., 2020. Animal carcass- and wood-derived biochars improved nutrient bioavailability, enzyme activity, and plant growth in metal-phthalic acid ester co-contaminated soils: A trial for reclamation and improvement of degraded soils. J. Environ. Manage. 261, 110246. https://doi.org/10.1016/j.jenvman.2020.110246.

Coors, A., Edwards, M., Lorenz, P., Römbke, J., Schmelz, R.M., Topp, E., Waszak, K., Wilkes, G., Lapen, D.R., 2016. Biosolids applied to agricultural land: Influence on structural and functional endpoints of soil fauna on a short- and long-term scale. Sci. Total Environ. 562, 312-326. https://doi.org/10.1016/j.scitotenv.2016.03.226.

Corradini, F., Meza, P., Eguiluz, R., Casado, F., Huerta-Lwanga, E., Geissen, V., 2019. Evidence of microplastic accumulation in agricultural soils from sewage sludge disposal. Sci. Total Environ. 671, 411-420. https://doi.org/10.1016/j.scitotenv.2019.03.368.

Crossman, J., Hurley, H.H., Futter, M., Nizzetto, L., 2020. Transfer and transport of microplastics from biosolids to agricultural soils and the wider environment. Sci. Total Environ. 724, 138334. https://doi.org/10.1016/j.scitotenv.2020.138334.

de Souza Machado, A.A., Lau, C.W., Till, J., Kloas, W., Lehmann, A., 2018. Impacts of microplastics on the soil biophysical environment. Environ. Sci. Technol. 52, 9656-9665. https://doi.org/10.1021/acs.est.8b02212.

de Souza Machado, A.A., Lau, C.W., Kloas, W., Bergmann, J., Bacheher, J.B., Faltin, E., Becker, R., Goerlich, A.S., Rillig, M.C., 2019. Microplastics can change soil properties and affect plant performance. Environ. Sci. Technol. 53, 6044-6052. https://doi.org/10.1021/acs.est.9b01339.

Dempsey, M.A., Fisk, M.C., Yavitt, J.B., Fahey, T.J., Balser, T.C., 2013. Exotic earthworms alter soil microbial community composition and function. Soil Biol. Biochem. 67, 263-270. https://doi.org/10.1016/j.soilbio.2013.09.009.

Dong, L., Tang, X.Q., Lin, L., Li, C., Li, R., Wu, M., 2018. Pollution characteristics and source identification of polycyclic aromatic hydrocarbons and phthalic acid esters during high water level periods in the Wuhan section of the Yangtze River, China. Environ. Sci. 39(6), 2588-2599. https://doi.org/10.13227/j.hjkx.201710014.

Dong, Y.M., Gao, M.L., Song, Z.G., Qiu, W., 2020. Microplastic particles increase arsenic toxicity to rice seedlings. Environ. Pollut. 259, 113892. https://doi.org/10.1016/j.envpol.2019.113892.

Dris, R., Gasperi, J., Rocher, V., Saad, M., Renault, N., Tassin, B., 2015. Microplastic contamination in an urban area: a case study in Greater Paris. Environ. Chem. 12(5), 592-599. https://doi.org/10.1071/EN14167.

Duan, J., Bolan, N., Li, Y., Ding, S., Atugoda, T., Vithanage, M., Kirkham, M.B., 2021. Weathering of microplastics and interaction with other coexisting constituents in terrestrial and aquatic environments. Water Res. 196(4). https://doi.org/10.1016/j.watres.2021.117011.

Durgesh, T.K., Shweta., Singh, S., Swati, S., Rishikesh, P., Vijay, P.S., Nilesh, C.S., Sheo, M.P., Nawal, K.D., Devendra, K.C., 2016. An overview on manufactured nanoparticles in plants: Uptake, translocation, accumulation and phytotoxicity. Plant Physiol. Bioch. 110, 2-12. https://doi.org/10.1016/j.plaphy.2016.07.030.

Edwards, C.A., 2004. Earthworm Ecology. https://doi.org/10.1007/978-94-009-5965-1_14.

European Chemicals Agency (ECHA), 2019. Annex XV Restriction Report. Proposal for a Restriction. Intentionally Added Microplastics; ECHA: Helsinki, Finland. pp. 1-3.

European Commission (DG Environment), 2017. Intentionally added microplastics in products; Final Report. European Commission (DG Environment): London, UK. pp. 39-78.

Fakour, H., Lo, S.L., Yoashi, N.T., Massao, A.M., Lema, N.N., Mkhontfo, F.B., Jomalema, P.C., Jumanne, N.S., Mbuya, B.H., Mtweve, J.T., Imani, M., 2021. Quantification and analysis of microplastics in farmland soils: characterization, sources, and pathways. Agri. 11(4), 330. https://doi.org/10.3390/agriculture11040330.

Fei, Y., Huang, S., Zhang, H., Tong, Y., Barceló, D., 2019. Response of soil enzyme activities and bacterial communities to the accumulation of microplastics in an acid cropped soil. Sci. Total Environ. 707, 135634. https://doi.org/10.1016/j.scitotenv.2019.135634.

Fierer, N., Bradford, M.A., Jackson, R.B., 2007. Toward an ecological classification of soil bacteria. Ecology 88 (6), 1354-1364. https://doi.org/10.1890/05-1839.

Fincheira, P., Quiroz, A., 2018. Microbial volatiles as plant growth inducers. Microbiol. Res. 208, 63-75. https://doi.org/10.1016/j.micres.2018.01.002.

Fischer, E.K., Paglialonga, L., Czech, E., Tamminga, M., 2016. Microplastic pollution in lakes and lake shoreline sediments – A case study on Lake Bolsena and Lake Chiusi (Central Italy). Environ. Pollut. 213, 648-657. https://doi.org/10.1016/j.envpol.2016.03.012.

Fuller, S., Gautam, A., 2016. A procedure for measuring microplastics using pressurized fluid extraction. Environ. Sci. Technol. 50, 5774-80. https://doi.org/10.1021/acs.est. 6b00816.

Gao, Y., Collins, C.D., 2009. Uptake Pathways of Polycyclic Aromatic Hydrocarbons in White Clover. Environ. Sci. Technol. 43, 6190-5. https://doi.org/10.1021/es900662d.

Gao, M.L., Liu, Y., Song. Z.G., 2019. Effects of polyethylene microplastic on the phytotoxicity of di-n-butyl phthalate in lettuce (Lactuca sativa L. var. ramosa Hort). Chemosphere 237, 124482. https://doi.org/10.1016/j.chemosphere.2019.124482.

Gao, M.L., Liu, Y., Dong, Y., Song, Z.G., 2020. Effect of polyethylene particles on dibutyl phthalate toxicity in lettuce (Lactuca sativa L.). J. Hazard. Mater. 401, 123422. https://doi.org/10.1016/j.jhazmat.2020.123422.

Geyer, R., Jambeck, J.R., Law, K.L., 2017. Production, use, and fate of all plastics ever made. Sci. Adv. 3, e1700782. https://doi.org/10.1126/sciadv.1700782.

Godoy, V., Blazquez, G., Calero, M., Quesada, L., Martin-Lara, M.A., 2019. The potential of microplastics as carriers of metals. Environ. Pollut. 255, 113363.1-113363.12. https://doi.org/10.1016/j.envpol.2019.113363.

Harms, I.K., Diektter, T., Troegel, S., Lenz, M., 2020. Amount, distribution and composition of large microplastics in typical agricultural soils in Northern Germany. Sci. Total Environ. 758, 143615. https://doi.org/10.1016/j.scitotenv.2020.143615.

Harrison, J.P., Schratzberger, M., Sapp, M., Osborn, A., 2014. Rapid bacterial colonization of low-density polyethylene microplastics in coastal sediment microcosms. BMC Microbiol. 14, 232. https://doi.org/10.1186/s12866-014-0232-4.

Hartmann, N.B., Rist, S., Bodin, J., Jensen, L.H., Schmidt, S.N., Mayer, P., Meibom, A., Baun, A., 2017. Microplastics as vectors for environmental contaminants: Exploring sorption, desorption, and transfer to biota. Integr. Environ. Assess. Manag. 13, 488-493. https://doi.org/10.1002/ieam.1904.

Hoang, D.T.T., Razavi, B.S., Kuzyakov, Y., Blagodatskaya, E., 2016. Earthworm burrows: kinetics and spatial distribution of enzymes of C-, N- and P- cycles. Soil Biol. Biochem. 99, 94-103. https://doi.org/ 10.1016/j.soilbio.2016.04.021.

Hodson, M.E., Duffus-Hodson, C.A., Clark, A., Prendergast-Miller, M.T., Thorpe, K.L., 2017. Plastic bag derived-microplastics as a vector for metal eExposure in terrestrial invertebrates. Environ. Sci. Technol. 51, 4714-4721. https://doi.org/10.1021/acs.est.7b00635.

Hortal, S., Bastida, F., Armas, C., Lozano, Y.M., Moreno, J.L., García, C., Pugnaire, F.I., 2013. Soil microbial community under a nurse-plant species changes in composition, biomass and activity as the nurse grows. Soil Biol. Biochem. 64, 139-146. https://doi.org/10.1016/j.soilbio.2013.04.018.

Horton, A.A., Walton, A., Spurgeon, D.J., Lahive, E., Svendsen, C., 2017. Microplastics in freshwater and terrestrial environments: Evaluating the current understanding to identify the knowledge gaps and future research priorities. Sci. Total Environ. 586, 15. https://doi.org/10.1016/j.scitotenv.2017.01.190.

Huang, Y., Zhao, Y., Wang, J., Zhang, M., Jia, W., Qin, X., 2019. LDPE microplastic films alter microbial community composition and enzymatic activities in soil. Environ. Pollut. 254(Pt A), 112983. https://doi.org/10.1016/j.envpol.2019.112983.

Huerta, L.E., Gertsen, H., Gooren, H., Gooren, H.P.A., Peters, P., Salánki, T., van der Ploeg, M.J., Besseling, E., Koelmans, A., Geissen, V., 2016. Microplastics in the Terrestrial Ecosystem: Implications for Lumbricus terrestris (Oligochaeta, Lumbricidae). Environ. Sci. Technol. 50(5), 2685. https://doi.org/10.1021/acs.est.5b05478.

Hüffer, T., Hofmann, T., 2016. Sorption of non-polar organic compounds by micro-sized plastic particles in aqueous solution. Environ. Pollut. 214, 194-201. https://doi.org/10.1016/j.envpol.2016.04.018.

Jiang, X., Chen, H., Liao, Y., Ye, Z., Li, M., Klobuar, G., 2019. Ecotoxicity and genotoxicity of polystyrene microplastics on higher plant Vicia faba. Environ. Pollut. 250, 831-838. https://doi.org/10.1016/j. envpol.2019.04.055.

Kalčíková, G., Alič, B., Skalar, T., Bundschuh, M., Gotvajn, A.Ž., 2017a. Wastewater treatment plant effluents as source of cosmetic polyethylene microbeads to freshwater. Chemosphere 188, 25-31. https://doi.org/10.1016/j.chemosphere.2017.08.131.

Kalčíková, G., Gotvajna, A.Z., Kladnik, A., Jemec, A., 2017b. Impact of polyethylene microbeads on the floating freshwater plant duckweed Lemna minor. Environ. Pollut. 230, 1108-1115. https://doi.org/10.1016/j.envpol.2017.07.050.

Kasirajan, S., Ngouajio, M., 2012. Polyethylene and biodegradable mulches for agricultural applications: A review. Agron. Sustain. Dev. 32(2), 501-529. https://doi.org/10.1007/s13593-011-0068-3.

Kim, M.N., Shin, J.H., Im, S.S, 2003. Effect of poly (l-Lactide) and poly (Butylene Succinate) on the growth of red pepper and tomato. J. Polym. Environ. 11(3), 101-105. https://doi.org/10.1023/A:1024683014153.

Kirkham, M.B, 2020. Water relations and cadmium uptake of wheat grown in soil with particulate plastics. pp.195. https://doi.org/10.1201/9781003053071-16.

Koelmans, A.A., Besseling, E., Shim, W.J., 2015. Nanoplastics in the Aquatic Environment. Critical Review. Marine Anthropogenic Litter. Springer.

Kumar, M., Xiong, X., He, M., Tsang, D.C.W., Bolan, N.S., 2020. Microplastics as pollutants in agricultural soils. Environ. Pollut. 265(Pt A), 114980. https://doi.org/10.1016/j.envpol.2020.114980.

Lee, H., Kim, Y., 2018. Treatment characteristics of microplastics at biological sewage treatment facilities in Korea. Mar Pollut. Bull. 137, 1-8. https://doi.org/10.1016/j.marpolbul.2018.09.050.

Li, X.W., Chen, L.B., Mei, Q.Q., Dong, B., Dai, X., Ding, G., Dai, X.H., 2018a. Microplastics in sewage sludge from the wastewater treatment plants in China. Water Res. 142, 75-85. https://doi.org/10.1016/j.watres.2018.05.034.

Li, J., Zhang, K., Zhang, H., 2018b. Adsorption of antibiotics on microplastics. Environ. Pollut. 237, 460-467. https://doi.org/10.1016/j.envpol.2018.02.050.

Li, L.Z., Zhou, Q., Yin, N., Zhang, Y., 2019. Uptake and accumulation of microplastics in an edible plant. Chinese Journal 64, 928-934. https://doi.org/10.1360/N972018-00845.

Li, W.F., Wufuer, R., Duo, J., Wang, S.Z., Luo, Y.M., Zhang, D.Y., Pan, X.L., 2020a. Microplastics in agricultural soils: Extraction and characterization after different periods of polythene film mulching in an arid region. Sci. Total Environ. 749(5672), 141420. https://doi.org/10.1016/j.scitotenv.2020.141420.

Li, L.Z., Luo, Y.M., Li, R.J., Zhou, Q., Zhang, Y., 2020b. Effective uptake of submicrometre plastics by crop plants via a crack-entry mode. Nat. Sustain. 3, 929-937. https://doi.org/10.1038/s41893-020-0567-9.

Li, Z.X., Li, Q.F., Li, R.J., Wang, G., 2020c. Physiological responses of lettuce (Lactuca sativa L.) to microplastic pollution. Sci. Pollut. Res. (2), 1-9. https://doi.org/10.1007/s11356-020-09349-0.

Liu, F.F., Liu, Z.Z., Zhu, Z.L., Wang, S.C., Zhao, F.F., 2018a. Interactions between microplastics and phthalate esters as affected by microplastics characteristics and solution chemistry. Chemosphere 214, 688-694. https://doi.org/10.1016/j.chemosphere2018.09.174.

Liu, M., Lu, S., Song, Y., Lei, L., Hu, J., Lv, W., Zhou, W., Cao, C., Shi, H., Yang, X., 2018b. Microplastic and mesoplastic pollution in farmland soils in suburbs of Shanghai, China. Environ. Pollut. 242, 855-862. https://doi.org/10.1016/j.envpol.2018.07.051.

Liu, K., Wang, X., Song, Z., Wei, N., Li, D., 2020. Terrestrial plants as a potential temporary sink of atmospheric microplastics during transport. Sci. Total Environ. 742, 140523. https://doi.org/10.1016/j.scitotenv.2020.140523.

Lozano, Y.M., Lehnert, T., Linck, L.T., Lehmann, A., Rillig, M.C., 2020. Microplastic shape, concentration and polymer type affect soil properties and plant biomass. https://doi.org/10.1101/2020.07.27.223768.

Lozano, Y.M., Rillig, M.C., 2020. Effects of microplastic fibers and drought on plant communities. Environ. Sci. Technol. 54, 6166-6173. https://doi.org/10.1021/acs.est.0c01051.

Lv, W.W., Wen, Z.Z., Lu, S.B., Huang, W.W., Yuan, Q., Tian, M.L., Lv, W.G., He, D.F., 2019. Microplastic pollution in rice-fish co-culture system: A report of three farmland stations in Shanghai, China. Sci. Total Environ. 652, 1209-1218. https://doi.org/10.1016/j.scitotenv.2018.10.321.

Mao, Y.F., Ai, N., Chen, Y., Zhang, Z., Zeng, P., Kang, L., Li, W., Gu,W.K., He, Q., Li, H., 2018. Phytoplankton response to polystyrene microplastics: perspective from an entire growth period. Chemosphere 208, 59-68. https://doi.org/10.1016/j.chemosphere.2018.05.170.

Martin-Closas, L., Botet, R., Pelacho, A.M., 2014. An in vitro crop plant ecotoxicity test for agricultural Bioplastic constituents. Polym. Degrad. Stabil. 108, 250-256. https://doi.org/10.1016/j.polymdegradstab.2014.03.037.

Menéndez-Pedriza, A., Jaumot, J., 2020. Interaction of environmental pollutants with microplastics: a critical review of sorption factors, bioaccumulation and ecotoxicological Effects. Toxics 8(2), 1-39. https://doi.org/10.3390/toxics8020040.

Miller, E.L., Nason, S.L., Karthikeyan, K.G., Pedersen, J.A., 2016. Root Uptake of pharmaceuticals and personal care product ingredients. Environ. Sci. Technol. 50(2), 525-41. https://doi.org/10.1021/acs.est.5b01546.

Nizzetto, L., Futter, M., Langaas, S., 2016. Are agricultural soils dumps for microplastics of urban origin? Environ. Sci. Technol. 50, 10777-10779. https://doi.org/10.1021/acs.est.6b04140.

Paré, P.W., Tumlinson, J.H., 1999. Plant volatiles as a defense against insect herbivores. Plant Physiol. 121(2), 325-331. https://doi.org/10.1104/pp.121.2.325.

Patterson, J., Jeyasanta, K.I., Sathish, N., Edward, J.K.P., Booth, A.M., 2020. Microplastic and heavy metal distributions in an Indian coral reef ecosystem. Sci. Total Environ. 744, 140706. https://doi.org/10.1016/j.scitotenv.2020.140706.

Piehl, S., Leibner, A., Martin, G.J.L., Dris, R., Bogner, C., Laforsch, C., 2018. Identification and quantification of macro- and microplastics on an agricultural farmland. Sci. Rep. 8(1), 17950. https://doi.org/10.1038/s41598-018-36172-y.

Pietra, L.D, 2019. The EU`s push for biodegradable polymers in mineral fertilizers, fertilizer focus; Fertilizers Europe: Brussels, Belgium. pp. 1-2.

Poeta, G., Fanelli, G., Pietrelli, L., Acosta, A.T.R., Battisti, C., 2017. Plastisphere in action: evidence for an interaction between expanded polystyrene and dunal plants. Environ. Sci. Pollut. Res. 24(12), 11856-11859. https://doi.org/10.1007/s11356-017-8887-7.

Qi, Y.L., Yang, X.M., Pelaez, A.M., Esperanza, H.L., Nicolas, B., Henny, G., Paolina, G., Violette, G., 2018. Macro- and micro- plastics in soil-plant system: Effects of plastic mulch film residues on wheat (Triticum aestivum) growth. Sci. Total Environ. 645, 1048-1056. https://doi.org/10.1016/j.scitotenv.2018.07.229.

Ramos, L., Berenstein, G., Hughes, E.A., Zalts, A., Montserrat, J.M., 2015. Polyethylene film incorporation into the horticultural soil of small periurban production units in Argentina. Sci. Total Environ. 523(aug.1), 74-81. https://doi.org/10.1016/j.scitotenv.2015.03.142.

Rice-Evans, C., Miller, N., Paganga, G., 1997. Antioxidant properties of phenolic compounds. Trends Plant Sci. 2(4), 0-159. https://doi.org/10.1016/s1360-1385(97)01018-2.

Rillig, M.C., 2012. Microplastic in terrestrial ecosystems and the soil. Environ. Sci. Technol. 46, 6453-4. https://doi.org/10.1021/es302011r.

Rillig, M.C., Ziersch, L., Hempel, S., 2017. Microplastic transport in soil by earthworms. Sci. Rep. 7, 1362. https://doi.org/10.1038/s41598-017-01594-7.

Rillig, M.C., 2018. Microplastic disguising as soil carbon storage. Environ. Sci. Technol. 52, 6079. https://doi.org/10.1021/acs. est.8b02338.

Rillig, M.C., Lehmann, A., Machado, A.A.D.S., Yang, G., 2019. Microplastic effects on plants. New Phytol. 223(3), 1066-1070. https://doi.org/0000-0003-3541-7853.

Rillig, M.C., 2020. Plastic and plants. Nat. Sustain. 3(11), 1-2. https://doi.org/10.1038/s41893-020-0583-9.

Saglam, M., Sintim, H.Y., Bary, A.I., Miles, C.A., Ghimire, S., Inglis, D.A., Flury, M., 2017. Modeling the effect of biodegradable paper and plastic mulch on soil moisture dynamics. Agri.Water Manag. 193, 240–250. https://doi.org/10.1016/j.agwat.2017.08.011.

Sanchez-Hernandez, J.C., Capowiez, Y., Ro, K.S., 2020. Potential use of earthworms to enhance decaying of biodegradable plastics. ACS Sustain. Chem. Eng. 8, 4292-4316. https://doi.org/10.1021/acssuschemeng.9b05450.

Schreiber, L., Hartmann, K., Skrabs, M., Zeier, J., 1999. Apoplastic barriers in roots: chemical composition of endodermal and hypodermal cell walls. J. Exp. Bot. (337), 1267-1280. https://doi.org/10.1093/jexbot/50.337.1267.

Sighicelli, M., Pietrelli, L., Lecce, F., Iannilli, V., Falconieri, M., Coscia, L., Vito, S.D., Nuglio, S., Zampetti, G., 2018. Microplastic pollution in the surface waters of Italian Subalpine Lakes. Environ. Pollut. 236, 645-651. https://doi.org/10.1016/j.envpol.2018.02.008.

Smith, H., 1978. The molecular biology of plant cells. Bull. Torrey Bot. Club. 105(4). https://doi.org/10.2307/2484929.

Song, Y.K., Sang, H.H., Mi, J., Han, G.M., Jung, S.W., Shim, W.J., 2018. Corrections to "Combined effects of UV exposure duration and mechanical abrasion on microplastic fragmentation by polymer type". Environ. Sci. Technol. 52(6), 4368-4376. https://doi.org/10.1021/acs.est.8b00172.

Souza, P.M.S., Corroque, N.A., Morales, A.R., Mei, L.H.I., 2013. PLA and organoclays nanocomposites: Degradation process and evaluation of ecotoxicity using *Allium cepa* as test organism. J. Polym. Environ. 21, 1052-1063. https://doi.org/10.1007/s10924-013-0604-0.

Sridharan, S., Kumar, M., Bolan, M.S., Singh, L., Kumar, S., Kumar, R., You, S., 2021. Are microplastics destabilizing the global network of terrestrial and aquatic ecosystem services? Environ. Res.198, 111243. https://doi.org/ 10.1016/j.envres.2021.111243.

Staniszewska, M., Graca., Nehring, I., 2016. The fate of bisphenol A, 4-tert-octylphenol and 4-nonylphenol leached from plastic debris into marine water -experimental studies on biodegradation and sorption on suspended particulate matter and nano-TiO_2. Chemosphere 145, 535-542. https://doi.org/10.1016/j.chemosphere.2015.11.081.

Steinmetz, Z., Wollmann, C., Schaefer, M., Buchmann, C., David, J., Tröger, J., Muñoz, K., Frör, O., Schaumann, G.E., 2016. Plastic mulching in agriculture. Trading short-term agronomic benefits for long-term soil degradation. Sci. Total Environ. 550, 690-705. https://doi.org/10.1016/j.scitotenv.2016.01.153.

Stubenrauch, J., Ekardt, F., 2020. Plastic pollution in soils: governance approaches to foster soil health and closed nutrient cycles. Environ. 7(5), 38. https://doi.org/10.3390/environments7050038.

Sun, X.D., Yuan, X.Z., Jia, Y., Feng, L.J., Xing, B., 2020. Differentially charged nanoplastics demonstrate distinct accumulation in Arabidopsis thaliana. Nat. Nanotechnol. 15, 755-760. https://doi.org/10.1038/s41565-020-0707-4.

Taylor, S.E., Pearce, C.I., Sanguinet, K.A., Hu, D., Chrisler, W.B., Kim, Y.M., Wang, Z., Flury, M., 2020. Polystyrene nano- and microplastic accumulation at Arabidopsis and wheat root cap cells, but no evidence for uptake into roots. Environ. Sci. Nano. 7(7). https://doi.org/10.1039/D0EN00309C.

Teuten, E.L., Saquing, J.M., Knappe, D.R., Barlaz, M.A., Jonsson, S., Björn, A., Rowland, S.J., Thompson, R.C., Galloway, T.S., Yamashita, R., 2009. Transport and release of chemicals from plastics to the environment and to wildlife. Philos. Trans: Biol. Sci. 364(1526), 2027-2045. https://doi.org/10.1098/rstb.2008.0284.

van den Berg, P., Huerta-Lwanga, E., Corradini, F., Geissen, V., 2020. Sewage sludge application as a vehicle for microplastics in eastern Spanish agricultural soils. Environ. Pollut. 261, 114198. https://doi.org/10.1016/j.envpol.2020.114198.

Van Sebille, E., Wilcox, C., Lebreton, L., Maximenko, N., 2015. A global inventory of small floating plastic debris. Environ. Res. Lett. 10, 124006. https://doi.org/10.1088/1748-9326/10/12/124006.

Van Weert, S., Redondo-Hasselerharm, P.E., Diepens, N.J., Koelmans, A.A., 2018. Effects of nanoplastics and microplastics on the growth of sediment-rooted macrophytes. Sci. Total Environ. 654, 1040-1047. https://doi.org/10.1016/j.scitotenv.2018.11.183.

Wallace, H., 2016. Presence of microplastics and nanoplastics in food, with particular focus on seafood. EFSA Journal 14(6), 4501. https://doi.org/10.2903/j.efsa.2016.4501.

Waller, L.P., Allen, W.J., Barratt, P., Condron, L.M., Dickie, I.A., 2020. Biotic interactions drive ecosystem responses to exotic plant invaders. Sci. (New York, N.Y.). 368(11), 5774-5780. https://doi.org/10.1126/science.aba2225.

Wang, C.R., Tian, Y., Wang, X., Geng, J., Jiang, J., Yu, H., Chen, W., 2010. Lead-contaminated soil induced oxidative stress, defense response and its indicative biomarkers in roots of Vicia faba seedlings. Ecotoxicol. 19, 1130-1139. https://doi.org/10.1007/s10646-010-0496-x.

Wang, Z., Qin, Y., Li, W., Yang, W., Yang, J., 2019. Microplastic contamination in freshwater: first observation in Lake Ulansuhai, Yellow River Basin, China. Environ. Chem. Lett. 17, 1821-1830. https://doi.org/10.1007/s10311-019-00888-8.

Wang, T.T., Ying, G.G., He, L.Y., Liu, Y.S., Zhao, J.L., 2020a. Uptake mechanism, subcellular distribution, and uptake process of perfluorooctanoic acid and perfluorooctane sulfonic acid by wetland plant Alisma orientale. Sci. Total Environ. 733, 139383. https://doi.org/10.1016/j.scitotenv.2020.139383.

Wang, F., Zhang, X., Zhang, S., Zhang, S., Sun, Y., 2020b. Interactions of microplastics and cadmium on plant growth and arbuscular mycorrhizal fungal communities in an agricultural soil. Chemosphere 254, 126791. https://doi.org/10.1016/j.chemosphere.2020.126791.

Wang, X., Bolan, N., Tsang, D. C. W., Sarkar, B., Bradney, L., Li, Y., 2021a. A review of microplastics aggregation in aquatic environment: influence factors, analytical methods, and environmental implications. J. Hazard. Mater. 402, 19. https://doi.org/10.1016/j.jhazmat.2020.123496.

Wang, L.W., Wu, W.M., Bolan, N.S., Dan, T., 2021b. Environmental fate, toxicity and risk management strategies of nanoplastics in the environment: Current status and future perspectives. J. Hazard. Mater. 401, 123415. https://doi.org/10.1016/j.jhazmat.2020.123415.

Weithmann, N., Möller, J.N., Löder, M.G.J., Piehl, S., Laforsch, C., Freitag. R., 2018. Organic fertilizer as a vehicle for the entry of microplastic into the environment. Sci. Adv. 4(4), eaap8060. https://doi.org/10.1126/sciadv.aap8060.

Wu, X., Liu, Y., Yin, S., Xiao, K., Yang, J., 2020. Metabolomics revealing the response of rice (Oryza sativa L.) exposed to polystyrene microplastics. Environ. Pollut. 266, 115159. https://doi.org/10.1016/j.envpol.2020.115159.

Yi, M., Zhou, S., Zhang, L., Ding, S., 2020. The effects of three different microplastics on enzyme activities and microbial communities in soil. Water Environ. Res. 93(1), 24-32. https://doi.org/10.1002/wer.1327.

Yoshihisa, K., Yoshimura, A., Shibamori, Y., Fuchigami, K., Kubota, N., 2012. Polymer surface modification by using microwave plasma irradiation. J. Solid Mech. Mater. Eng. 6, 654-659. https://doi.org/10.1299/jmmp.6.654.

Zang, H., Zhou, J., Marshall, M.R., Chadwick, D.R., Wen, Y., Jones, D.L., 2020. Microplastics in the agroecosystem: Are they an emerging threat to the plant-soil system. Soil Biol. Biochem. 148, 107926. https://doi.org/10.1016/j.soilbio.2020.107926.

Zhang, B., Chu, G.X., Wei, C.Z., Ye, J., 2011. The growth and antioxidant defense responses of wheat seedlings to omethoate stress. Pestic. Biochem. Physiol. 100, 273-279. https://doi.org/10.1016/j.pestbp.2011.04.012.

Zhang, C., Chen, X., Wang, J., Tan, L., 2016. Toxic effects of microplastic on marine microalgae Skeletonema costatum: Interactions between microplastic and algae. Environ. Pollut. 220, 1282-1288. https://doi.org/10.1016/j.envpol.2016.11.005.

Zhang, G.S., Liu, Y.F., 2018. The distribution of microplastics in soil aggregate fractions in southwestern China. Sci. Total Environ. 642, 12-20. https://doi.org/10.1016/j.scitotenv.2018.06.004.

Zhang, H.L., Lu, L., Zhao, X.P., Zhao, S., Gu, X.Y., Du, W.C., Wei, H., Ji, R., Zhao, L.J., 2019. Metabolomics reveal the "invisible" responses of Spinach plants exposed to CeO_2 nanoparticles. Environ. Sci. Technol. 53, 6007-6017. https://doi.org/10.1021/acs.est.9b00593.

Zhang, D., Ng, E.L., Hu, W., Wang, H.Y., 2020a. Plastic pollution in croplands threatens long-term food security. Glob. Chang. Biol. 26, 3356-3367. https://doi.org/10.1111/gcb.15043.

Zhang, L.S., Xie, Y.S., Liu, J.Y., 2020b. An overlooked entry pathway of microplastics into agricultural soils from application of sludge-based fertilizers. Environ. Sci. Technol. 54 (7), 4248-4255. https://doi.org/10.1021/acs.est.9b07905.

Zhao, H.M., Huang, H.B., Du, H., Jing, L., Zhou, D.M., 2018. Intraspecific variability of ciprofloxacin accumulation, tolerance, and metabolism in Chinese flowering cabbage (brassica parachinensis). J. Hazard. Mater. 349, 252-261. https://doi.org/10.1016/j.jhazmat.2018.01.015.

Zhou, B., Wang, J., Zhang, H., Damià Barceló., 2019a. Microplastics in agricultural soils on the coastal plain of Hangzhou Bay, east China: Multiple sources other than plastic mulching film. J. Hazard. Mater. 388, 121814. https://doi.org/10.1016/j.jhazmat. 2019.121814.

Zhou, Y., Liu, X., Wang, J., 2019b. Characterization of microplastics and the association of heavy metals with microplastics in suburban soil of central China. Sci. Total Environ. 649, 133798.1-133798.10. https://doi.org/10.1016/j.scitotenv.2019.133798.

Zhou, Y., Liu, X., Wang, J., 2020. Ecotoxicological effects of microplastics and cadmium on the earthworm Eisenia foetida. J. Hazard. Mater. 392, 122273-. https://doi.org/10.1016/j.jhazmat. 2020. 122273.

Zhou, J., Wen, Y., Marshall, M.R., Zhao, J., Gui, H., Yang, Y.D., Zeng, Z.H., Jones, D.L., Zang, H.D., 2021. Microplastics as an emerging threat to plant and soil health in agroecosystems. Sci. Total Environ. 787, 147444. https://doi.org/10.1016/j.scitotenv.2021.147444.

Zhu, Z.L., Wang, S.C., Zhao, F.F., Wang, S.G., Liu, F.F., Liu, G. Z., 2019. Joint toxicity of microplastics with triclosan to marine microalgae Skeletonema costatum. Environ. Pollut. (Barking, Essex: 1987). 246(MAR.), 509-517. https://doi.org/10.1016/j.envpol.2018.12.044.

Figures

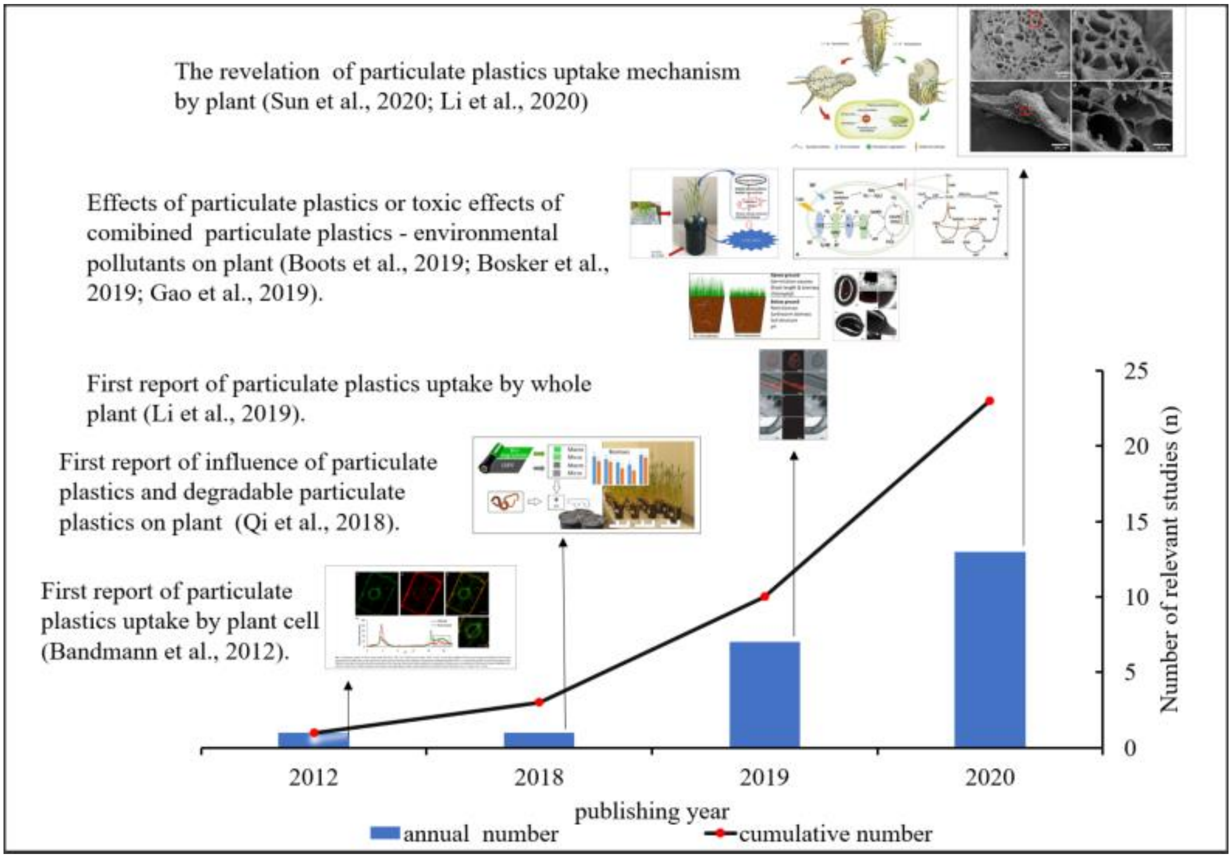


Fig. 1. Mapping the research progress on the interaction of particulate plastics with plants in recent years.

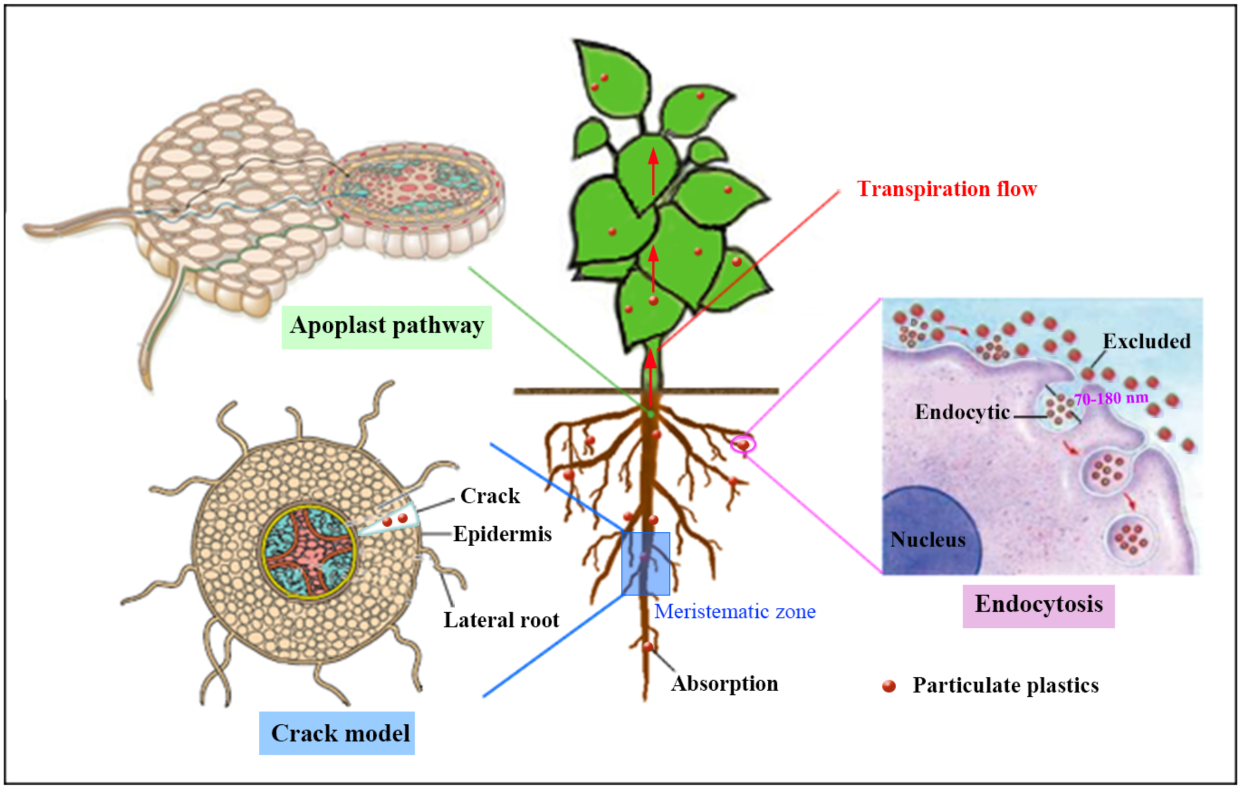


Fig. 2. Possible mechanisms of particulate plastics uptake by plants.

Table 1. Occurrence and characteristics of particulate plastics in agricultural soils.

Land use/crop type	Plastic abundance	Plastic size range	Plastic shape	Plastic type	Location	Reference
Wheat, barley, lucerne, triticale, white mustard, and corn	0.34 ± 0.36 particles/kg	2-4 mm (76.9%)	Film (43.75%), fragment (43.75%) and fiber (12.50%)	Polyethylene (62.5%), Polypropylene (25.0%), Polystyrene (12.5%)	Southeastern Germany	Piehl et al., 2018
Vegetable farm	7100- 42,960 particles/kg	1-0.05 mm (95%)	Fiber (92.1%), film (3.7%), fragment (4.1%), string (0.2%)	--	Yunnan, China	Zhang and Liu, 2018
Vegetable farm	320-12,5600 particles/kg	0.02-0.2 mm (70%), 0.2-0.5 mm (9%), 0.5-1.0 mm (13%)	Bead (48%), fiber (37%), fragment (15%), foam (1%)	Polyamide (32.5%), Polypropylene (28.8%), Polystyrene (16.9%), Polyvinyl chloride (1.9%), Polyethylene (4.2%)	Wuhan, China	Chen et al., 2019

Vegetable farm and orchard	10.10-61.05 mg/kg	0.9-2.0 mm	Fragment and fiber	--	Xinjiang, China	Li et al., 2020a
Vegetable farm	62.5-78 particles/kg	0.03-5 mm (93.3%), 5-16 mm (6.7%)	Fiber (53.33%), film (6.67%), fragment (37.58), and pellet (2.12%)	Polyethylene (43.43%), Polypropylene (50.51%), Polyester (6.06%)	Shanghai, China	Liu et al., 2018b
Vegetable farm and orchard	0-2760 particles/kg	0.06-3.5 mm	Film, fragment, fiber	Polyethylene, Polyamide, Polypropylene, Nylon, Polyester, Rayon, Acrylic	Shanghai, Hangzhou and Ningbo, China	Zhou et al., 2019a
Vegetable farm	8-540 mg/kg	0.1-5 mm (100%)	--	--	Loess plateau, China	Zhang and Liu 2018
Vegetable farm	4.3×10^4-6.2×10^5 particles/kg	< 50 μm (99.8%)	Fragment (59%), fiber (20.8%)	Polyethylene (36.1%), Polyamide (17.3%), Polypropylene (11.5%), and others (35.1%)	Wuhan, China	Zhou et al., 2019b

Cabbage, corn, asparagus, pumpkin, guava	12–117 particles /m^2	1-3 mm (65%), 3-5 mm (12%), <1 mm (15%), >5 mm (8%)	Fragment (43%), fiber (21%), foam (16%), film (13%), pellet (6%), microbead (0.6%), and others (0.4%)	Light-density Polyethylene, Polyethylene, Oxidized polyethylene, Polystyrene, Polypropylene	Taiwan, China	Fakour et al., 2021
Winter rapeseed, winter wheat, winter barley, silage maize, sugar beet, vegetable	0 to 217.8 particles/kg	--	Foil (61%), fragment (28%), platelet (10%)	Polyethylene (87%), Polypropylene (4%), Nylon(3%), Polyamide (3%)	Schleswig-Holstein, Northern Germany	Harms et al., 2020
Vegetable farm	2116 ±1024 particles/kg	--	--	--	Murcia, Southeast Spain	Beriot et al., 2021
Cereals	930 ± 740 light density particles/kg and 1100 ± 570 heavy density particles/kg	150 – 250 µm	Fragment (>90%)	Polypropylene, Polyvinylchloride	Valencia, East of Spain	van den Berg et al., 2020

--: Not reported

1239 **Table 2.** Particulate plastics uptake by various plant species.

Plant species	Plastic type	Plastic size	Uptake location	Reference
Wheat (*Triticum aestivum*) and lettuce (*Lactuca sativa*)	Polystyrene beads	0.2 and 2.0 μm	Root, stem and leaf	Li et al., 2020b
Lettuce (*Lactuca sativa*)	Polystyrene beads	0.2 μm	Root, stem and leaf	Li et al., 2019
Arabidopsis thaliana	Synthesized polystyrene particles	200 nm	Root	Sun et al., 2020
Tobacco BY-2 cells	Fluorescent nano-beads	20 nm	Turgescent and plasmolyzed cells	Bandmann et al., 2012
Cress (*Lepidium sativum*)	Green fluorescent plastic particles	20 and 40 nm	Root hairs and shoot	Bosker et al., 2019

1240

Table 3. Effects of particulate plastics on plants.

Plant species	Plastic type	Plastic concentration	Plastic size	Exposure time	Location	Effects	Reference
Vicia faba	Polystyrene	10, 50 and 100 mg/L	5 µm and 100 nm	48 hours	Root	Decreased length, fresh weight and dry weight; Increased the activities of superoxide dismutase and peroxidase by 5 µm polystyrene; Decreased the activities of catalase by 5 µm polystyrene; Increased the micronucleus frequency in cells by both size of polystyrene.	Jiang et al., 2019
Wheat *(Triticum aestivum)*	Light-density polyethylene	1% (w/w)	50 µm - 1 mm	2 months	Fruit	Decreased the number of fruits.	Qi et al., 2018
					Root	Significantly decreased biomass.	
					Leaf	Decreased numbers and area; Increased the relative chlorophyll content.	
	Biodegradable plastic		50 µm - 1mm		Tiller and fruit	Inhibited the plant height; Increased the tillers number; Decreased the number of fruits	
					Root	Significantly decreased biomass.	
					Shoot	Significantly decreased biomass.	

					Stem	Decreased diameter.	
					Leaf	Decreased areas and numbers; Increased the relative chlorophyll content	
Grasses (*Festuca brevipila*, *Holcus lanatus* and *Calamagrostis epigejos*) and herbs (*Achillea millefolium*, *Hieracium pilosella*, *Plantago lanceolata* and *Potentilla argentea*)	Polyester	0.4% (w/w)	30 µm	2 months	Shoot	Shoot mass increased by ~6 % and root mass by ~ 90%.	Lozano and Rillig., 2020
Perennial ryegrass *(Lolium perenne)*	Polylactic acid	0.1% (w/w)	0.6 - 363 µm	30 days	Seed	Decreased biomass. Inhibited seeds germination.	Boots et al., 2019
					Shoot	Decreased in shoot height; Increased the chlorophyll-a /chlorophyll-b ratio.	
	High-density Polyethylene		0.48 - 316 µm		Root	Increased the root biomass and the chlorophyll-a/chlorophyll-b ratio.	

Spring onions (*Allium fistulosum*)	Polyethylene high density	2% (w/w)	Average diameter 643 µm	1.5 months	Root	Increased length and area; Decreased average diameter.	de Souza Machado et al., 2019
	Polyethylene terephthalate		222 - 258 µm		Root	Increased length, area and the ratio between root and leaf dry biomass; Decreased average diameter.	
					Leaf	Decrease water content.	
	Polystyrene		547 - 555 µm		Root	Increased biomass, length and area; Decreased average diameter and tissue density.	
	Polyester	0.2% (w/w)	8 µm		Root	Increased biomass, length and area; Decreased average diameter and tissue density.	
					Leaf	Decrease water content and nitrogen content; Increased C-N ratio.	
	Polyamide	2% (w/w)	15 - 20 µm		Root	Increased length and area; Decreased average diameter, tissue density and the ratio between root	
					Leaf	Increased water content and nitrogen content; Decreased C-N ratio and leaf dry biomass.	
	Polypropylene		647 - 754		Root	Increased length, area and the ratio	

			µm			between root; Decreased average diameter.	
					Leaf	Increased dry biomass; Decrease water content.	
Myriophyllum spicatum	Polystyrene	3%~10% (w/w)	50 - 190 nm 20 - 500 µm	21days	Root	Increased dry weight.	Van Weert et al., 2018
					Shoot	Reduced main shoot length.	
Elodea sp			50 - 190 nm		Root	Increase in shoot and root biomass.	
					Shoot	Increase side shoot length.	
Arabidopsis thaliana	Negatively charged nanoplastics (PS -SO_3H)	0.3 and 1.0 g/kg	55 ± 6 nm	7 weeks	Root	Decreased primary root growth;	Sun et al., 2020
					Above-ground	Decreased 41.7% and 51.5% above -ground fresh weights;	
	Positively charged nanoplastics		71 ± 6 nm		Root	Decreased primary root growth.	
					Above-ground	Decreased the chlorophyll content.	
Mazie *(zea mas L.var.Wannuoyihao)*	Polyethylene	0.1%, 1% and 10% (w/w)	100-154 µm	1 month	Root	No significant effect on root biomass.	Wang et al., 2020
	Polylactic acid	10% (w/w)			Leaf	Decreased the chlorophyll content.	

Rice *(oryza sativa)*	Polystyrene	50, 250 and 500 mg/L	8.5-30.7 µm	21 days	Root	No significant effect on biomass and length.	Wu et al., 2020
					Shoot	Significantly decreased the biomass and length.	
					Leaf	Significantly reduced the activities of superoxide dismutase, peroxidase and malondialdehyde; Increased the activities of catalase and reactive oxygen species for exposure doses of 50 mg/l, while decreased for exposure doses of 250 and 500 mg/L.	
Lettuce *(Lactuca sativa* L.var.*romosa* Hort)	Polyethylene	0.25,0.5, 1.0 mg/mL	23 µm	14 and 28 days	Root	Significantly decreased the fresh and dry weights and length.	Gao et al., 2019
					Leaf	Significantly decreased the fresh and dry biomass and numbers.	
Carrot (*Daucus carota*)	Polypropylene, Polyester, Polyethylene, Polyamide, Polyethylene terephthalate, Polyurethane, Polycarbonate	0.1%~4 % (w/w)	< 5 mm	2 weeks	root	Increased the biomasses.	Lozano et al., 2020
					Shoot	Increased the biomasses.	