

# Phytoremediation of Microplastics: A Perspective on Its Practicality

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**ABSTRACT:** Microplastics have permeated all parts of the environment, rendering their removal essential. Numerous strategies ranging from the physical removal of mismanaged plastic items to the biodegradation of microplastics with microorganisms and biocatalysts have been proposed to alleviate microplastic pollution. Phytoremediation is one of the plastic-removing strategies, but it has not received much attention. This perspective paper aims to review the phytoremediation of microplastics and discuss its practicality. The paper shows that plants could act as interceptors and a temporary sink of microplastics by facilitating their deposition, adsorbing them, trapping them in the root zone, enabling them to cluster on the roots, taking them up, translocating them, and accumulating them in various plant parts. However, there was a lack of evidence pointing to the degradation of microplastics after they were adsorbed, taken up, and stored. Weak adsorption and environmental factors may cause the trapped microplastics to desorb, resuspend, or evade, thus also making plants a source of microplastics. The microplastics trapped and accumulated in plants may be transferred to the higher trophic levels of the food chain through ingestion and raise concerns over their ecotoxicities. Unlike localized pollution, microplastic pollution is widespread, which limits the applicability of phytoremediation. Besides, microplastics could adversely impact plant health and the ability of plants to remove other environmental pollutants. These drawbacks may reduce the attractiveness of phytoremediation unless it can be effectively combined with bioremediation to degrade microplastics.

**KEYWORDS:** Accumulate; adsorb; microplastics; phytoremediation; rhizosphere; translocation

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## 1. Introduction

Environmental pollution caused by microplastics (MPs) is no longer something new to the scientific community. Numerous publications on various facets of MPs, ranging from their occurrence, abundance, distribution, fates, transports, and interactions to their risks, removal, and substitution, have been made available [1–3]. MPs have been detected in air, water, and soil, making them a widespread concern. Their presence could be traced to the most remote ecosystems in the world, and they have also entered the food chain [4]. MPs have made their way into numerous food items [5, 6], for instance, commercial fish species, mussels, and crustaceans [7]. They have also been found in sugars, salts, fruits, and vegetables [8]. MPs

originate from the breakdown of large plastic items. These plastic items comprise municipal debris such as plastic bags, packaging, and plastic bottles that are improperly discarded, fishing equipment such as buoys and fishing nets, as well as farming films [9]. These macroplastics undergo photodegradation, partial biodegradation, and mechanical breakdown in the environment, resulting in the formation of smaller plastic particles of variable sizes. The fraction of plastic particles with sizes less than 5 mm is generally called microplastics. However, there have been attempts to better classify plastic particles according to their sizes since particles of smaller sizes are usually more abundant in the environment, and the current definition of MPs does not effectively distinguish small nano-sized particles [10]. Bermudez and Swarzenski, for instance, proposed to classify MPs based on the classification system used in plankton research universally. With the new classification, MPs fall in the size range of 1 to 1000  $\mu\text{m}$  [11]. MPs could also be discharged directly into the environment in the form of microbeads typically found in cleaning agents and personal care products, microfibers from the washing of synthetic textiles, and plastic pellets used for the manufacturing of plastic items [12].

A large proportion of MPs enters the air through the wearing of synthetic rubber tires due to their friction with road surfaces, the washing and drying of synthetic textiles, as well as the wear and tear of furniture finishings, walls, and ceilings [13]. The long-range transport of MPs by wind and ocean currents results in their permeation into some of the most remote regions in the world [4]. MPs were detected in the Southern Ocean at concentrations of 0–0.66 items/ $\text{m}^3$  and 0–1.96 items/ $\text{m}^3$  in surface and sub-surface waters, respectively [14]. With MPs detected widely in the environment, their removal becomes a priority. The reason is that MPs may harm living organisms in multiple ways. MPs may accumulate in living organisms, causing malnutrition, inflammation, growth retardation, and compromised reproductive health [15]. MPs were reported to induce oxidative stress and adversely affect liver and kidney functions in mice [16]. While it is uncertain how MPs could affect human health, a study using a human adenocarcinoma cell line found polystyrene (PS) MPs to be weakly embryotoxic [17]. Therefore, in addition to MPs removal, it is equally important to ensure plastic waste is properly reduced and managed and environmentally friendly plastic substitutes are developed to minimize the consumption of conventional plastics, hence the volume of waste generated [18, 19].

Various methods have been proposed to remove MPs from the environment. Plastic waste in terrestrial and aquatic environments could be physically or mechanically removed to reduce the subsequent formation of MPs from the degradation of plastic waste [20]. Membrane filtration has shown promising results in the removal of MPs from wastewater or drinking water treatment streams to produce treated water with a very low abundance or totally free of MPs. The common membrane filtration technologies employed are reverse osmosis, ultrafiltration, and nanofiltration. During membrane filtration, MPs are retained in the concentrate [21]. Airborne MP can also be removed through filtration, particularly with a high-efficiency particulate air (HEPA) filter that has a high removal efficiency [4]. Filtration, nonetheless, does not break down MPs. It only captures MPs and transfers or retains them on a medium that is usually easier to dispose of [22]. Therefore, biodegradation and enzyme-assisted degradation of MPs have been the preferred methods of MPs removal and have been subjected to extensive studies. Bacteria are popular candidates for MPs degradation. Those of the phylum Actinobacteria and the genera *Thermobifidia* and *Saccharomonospora* have demonstrated the

ability to degrade MPs [23]. *Pseudomonas* sp. TDA 1 strain was observed to degrade polyurethane. *Ideonella sakaiensis* is a classical plastic-degrading bacterium of the family Comamonadaceae capable of breaking down polyethylene terephthalate (PET) [23].

Furthermore, fungi are known to secrete enzymes such as cutinases, lipases, proteases, and lignocellulolytic enzymes, which can degrade plastics. *Fusarium solani*, *Aspergillus oryzae*, and *Humicola insolens* are examples of fungi that secrete cutinases [24]. *Humicola insolens* cutinase was observed to degrade 97% of low-crystallinity PET film by weight within 96 hours [25], while a cutinase from *Arxula adeninivorans* was capable of degrading polycaprolactone fiber mats [26]. Exposure of PET to lipase B from *Candida antarctica* led to its hydrolysis to terephthalic acid [27]. The common protease-producing fungi are those of the genera *Aspergillus*, *Rhizopus*, *Humicola*, *Thermoascus*, and *Thermomyces* [28]. There has been increasing interest in using plastic-degrading enzymes from microorganisms to degrade plastics through immobilization techniques. For instance, Schwaminger et al. immobilized PETase on superparamagnetic iron oxide nanoparticles, and the immobilized enzyme showed good recyclability and better enzymatic activity than free PETase in decomposing micro-PET [29]. Biotechnology has been employed to modify plastic-degrading enzymes to enhance their activity. An attempt was made with protein engineering to produce PETase mutants with enhanced affinity for PET and better-fitting active sites. The mutants exhibited higher PET-degrading activity than wild-type PETase, with up to a 2.5-fold increase in activity [30].

Research has also been launched into the ability of algae to degrade MP. Algae were observed to secrete exopolysaccharides and ligninolytic enzymes upon settling on the surface of plastics, which could facilitate the breakdown of plastics. Multiple algal species encompassing *Scenedesmus dimorphus*, *Anabaena spiroides*, *Navicula pupula*, and *Oscillatoria* sp. are able to colonize the surface of submerged plastics, especially polyethylene (PE) [31]. *Anabaena spiroides*, a blue-green alga, was able to degrade 8.18% of low-density polyethylene (LDPE) [32], while *Chlorella vulgaris* was shown to degrade bisphenol A, a common plastic additive, to a level below its detection limit [31]. In comparison to bioremediation, the use of phytoremediation for MPs removal is less popular, and it casts doubt on whether phytoremediation of MPs is only limited to algae, which are not true plants, and whether it is effective at all. This article aims to provide perspectives on the applicability of phytoremediation for MP removal and recommendations for its application. It has the novelty of explaining if the use of phytoremediation alone is effective in removing MPs from the environment since phytoremediation has been mentioned in literature as a potential strategy for this purpose.

## 2. Methods

This article is perspective-based, aiming to provide a brief yet critical insight into whether phytoremediation is practical for MPs removal from the environment [33]. It achieved the aim by examining relevant literature on phytoremediation of MPs published in the last 10 years with keywords comprising microplastics, plastics, phytoremediation, plants, biodegradation, etc., or a combination of the keywords for a refined search, such as phytoremediation of microplastics and degradation of plastics by plants. The search was conducted over established journal databases, namely Web of Science, Scopus, and ScienceDirect. Inclusion criteria for the articles retrieved are as below:

- The articles must have been published in the past 10 years.
- The articles must include or focus on the phytoremediation of MPs.
- The articles must state the efficiency of MPs biodegradation using plant species.
- The articles must focus on only true plants.

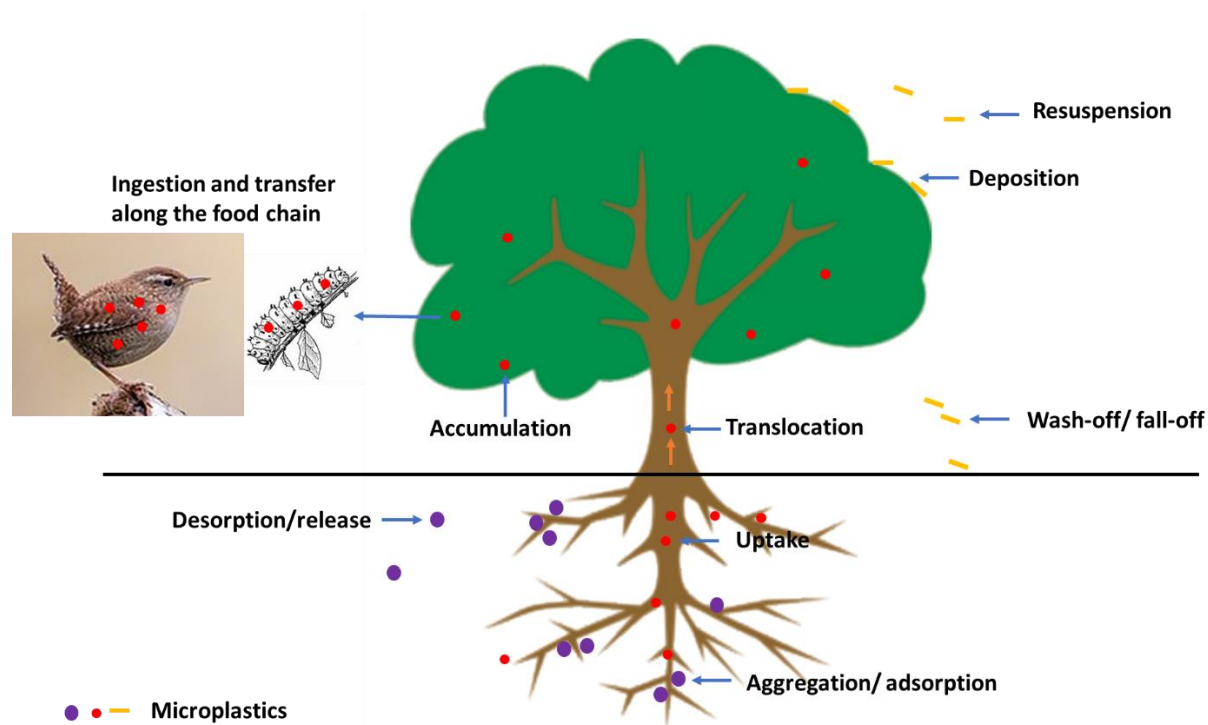
### 3. Results and Discussion

#### 3.1. Phytoremediation of MPs.

Numerous studies have discussed the potential of phytoremediation for the removal of MPs (Table 1). These studies can generally be divided into three types: the first examining the toxicity and distribution of MPs in or on plants intentionally exposed to MPs; the second studying the distribution and mechanisms of MPs translocation in plants; and the third looking into the effects of MPs on phytoremediation through co-exposure of plants to MPs and other pollutants. Exposure of *Lemna minor* (common duckweed) to PE MPs showed that the MPs adsorbed onto the surfaces of the plant in abundance, reaching 7 MPs per mm<sup>2</sup> with negligible impact on the plant [34] (Figure 1). Similarly, *Lemna minor* plants exposed to PE MP fragments were found to not have been negatively impacted in their growth, chlorophyll contents, or photosynthetic ability. The MPs adsorbed quickly on the plants, and approximately 6.5% of the adsorption was considered strong [35]. MPs were reported to alter the toxicity and availability of other pollutants. PE MPs reduced the uptake of dibutyl phthalate (DBP) by the roots of *Lactuca sativa* (lettuce) when present in a growth medium together, but they aggravated the toxic effect of DBP on roots once taken up. The PE MPs were observed to remain on root surfaces without entering the roots [36]. Exposing *Vicia faba* (fava bean) plants to 100 mg/L PS MPs for 48 hours resulted in the aggregation of PS MPs in their roots, which can interfere with nutrient transport [37]. A study on the spatial distribution of MPs in a mangrove habitat at Maowei, China, revealed a difference in the abundances of MPs in the rhizosphere and non-rhizosphere, with the former containing more MPs, thus indicating the intercepting effect of the rhizosphere on MPs [38]. MPs are retained by plants generally through rhizostabilization, during which MPs aggregate or adsorb on roots, and the nexus of roots in the rhizosphere tends to trap MPs (Figure 1). However, it is unclear if MPs could be effectively immobilized in the rhizosphere by rhizostabilization since Rozman et al. revealed that weak adsorption of MPs on roots is more predominant than strong adsorption (Table 1) and desorption could occur [35].

The interaction of plastic particles with roots is, in some instances, size-dependent. Giorgetti et al. observed the uptake of nanoPS by germinating *Allium cepa* (onions) and the ensuing accumulation of the nanoPS in different organelles of onion cells [39]. *Arabidopsis thaliana* (thale cress) plants exposed to functionalized PS nanoplastics were observed to accumulate PS nanoplastics [40]. This agrees with the findings of Giorgetti et al. on the uptake of nanoPS by onions [39] (Table 1). Aggregation of PS nanoplastics on roots was promoted by root exudates, which seem to inhibit the uptake of positively charged PS-NH<sub>2</sub>, suggesting the potential effect of surface charges of plastic particles on their uptake by roots [40]. In addition to size, the surface charges of plastic particles also determine their entry into plant roots. Another study on the effects of exposing *Lepidium sativum* (garden cress) seeds to nanoplastics and MP demonstrated the accumulation of the particles on root hairs (Table 1 and Figure 1). The study aimed primarily to examine how these plastic particles affected germination and did

not indicate if smaller plastic particles were taken up by the plants [41]. Helconski et al. conducted a study on the abundance and distribution of MPs in an urban tidal wetland, which revealed significant MPs accumulation in vegetated wetlands and that their spatial distribution was affected by hydrodynamics and vegetation [42]. This resonates with the findings of Li et al. that vegetation and its rhizosphere could intercept MPs and act as a temporary sink [38]. (Table 1). In addition to rhizosphere, the leaves of plants also serve as surfaces for the deposition of MPs, hence acting as interceptors and a temporary sink for MPs [43] (Figure 1). For smaller plastic particles, particularly nanoplastics, rhizofiltration is possible, where nanoplastics are removed from the environment through absorption by plant roots and the subsequent bioconcentration therein [44, 45]. This process could be affected by the surface charges of the plastic particles. Like MPs, nanoplastics can also cluster on roots and be rhizostabilized, during which they are immobilized or sequestered by roots [40].



**Figure 1.** Fates of microplastics during phytoremediation.

Studies have been conducted to track the transport of MPs in plants. Li et al. (2020) used fluorescent microbeads and a confocal laser scanning microscope to examine how PS beads were translocated in *Triticum aestivum* (wheat) [46]. They observed that the beads moved from roots to the aerial parts of the plant through the apoplastic pathway and the leaf veins, which implies that it is possible for plant roots to absorb MPs, which are subsequently transported to other parts of the plant [46] (Figure 1). This confirms phytoextraction or rhizofiltration (removal of contaminants by plants or roots from water bodies through root uptake or absorption) as a mechanism of MPs phytoremediation. A similar study involving the exposure of *Murraya exotica* (orange jasmine) to fluorescent poly(styrene-co-maleic anhydride) (SMA) nanoparticles and tracking their movement in the plant revealed translocation of the nanoparticles through the intercell-wall route in the epidermis, confirming the uptake of plastic nanoparticles and their translocation in plants [47]. The entry of MPs through roots was also shown by Austen et al. when they studied the uptake of polyamide (PA) MPs beads by *Betula*

*pendula* (silver birch) [48]. The accumulation of MPs in the young roots was affected by the size of the MPs, where those smaller than 10 m were found to be more abundant. This study again points to MPs size as a factor affecting their uptake by plants, and MPs of smaller sizes are usually taken up more easily. The translocation of MPs to other parts of the plant was not reported [48].

**Table 1.** Phytoremediation of MPs or nanoplastics and its implications.

Plant	Condition for MPs phytoremediation	Observation of MPs phytoremediation	Implication on phytoremediation	Reference
Leaves of plants in Shanghai and Lianyungang, China	Sampling of leaves to examine MPs deposition	Abundance of deposited MPs = 0.07 – 0.19 items/cm <sup>2</sup> of leaf.	Temporary sink of MPs where MPs could be released.	[43]
<i>Lemna minor</i> (L.)	Experimental plants were grown in Hutner's medium containing 50000 MPs/ml	PE sized 10 – 45 µm adsorbed strongly on the surfaces of the plant; each mm <sup>2</sup> of the plant adsorbed up to 7 PE MPs over 7 days; MPs adsorption had negligible impacts on photosynthesis and plant growth.	MPs adsorption facilitated transfer of MPs along the food chain.	[34]
<i>Lemna minor</i> (L.)	Exposure of plants to PE MPs fragments with an average size of 149 ± 75 µm at a concentration of 100 mg/L or 9600 MPs/L over 12 weeks	No negative effects on the growth, chlorophyll contents and photosynthetic activity of plants were observed; the roots developed tolerance to MPs; adsorption of MPs on the plants was rapid, leading to an average 6.5% and 20.0% strong and weak adsorption respectively.	MPs adhered on plants and were not degraded; saturation was reached by day-7; environmental conditions may alter MPs adsorption and promote their desorption.	[35]
<i>Lactuca sativa</i> L.	Exposure of DBP and PE to green and purple lettuces for 28 days separately or in combination	PE attached to the root surfaces without entering the roots; PE could interact with DBP to decrease its absorption by the roots; PE aggravated the toxic effect of DBP on lettuce roots	It is possible that PE of sizes smaller than lettuce root pores could enter the roots; desorption could happen and no degradation of MPs was reported.	[36]
<i>Triticum aestivum</i> L. Zhongmai 9	Exposure of experimental plants to microbeads stained with fluorescent dye, followed by imaging using confocal laser scanning microscope	Translocation of 0.2 µm PS beads from roots to the aboveground parts of the plants was observed; the translocation occurred via apoplastic pathway and the leaf veins.	MPs accumulated in plant with no sign of degradation; the MPs could transfer to animals through ingestion.	[46]
<i>Vicia faba</i>	Exposure of the root tips of <i>Vicia faba</i> to 10, 50 and 100 mg/L PS MPs sized 5 µm and 100 nm respectively for 48 hours	100 nm PS MPs gathered in the roots of <i>Vicia faba</i> and may hamper nutrient transport.	Accumulation of MPs could harm phytoremediating plants, causing ecological risk without apparent evidence of their biodegradation.	[37]
<i>Murraya exotica</i>	Exposure of <i>Murraya exotica</i> roots to fluorescent poly(styrene-co-maleic anhydride) nanoparticles followed by two-photon excitation and time-resolved optical imaging	SMA nanoparticles were taken up and translocated by the plant through an intercell-wall route in the epidermis.	MPs were retained in the plant.	[47]
<i>Allium cepa</i>	Exposure of germinating <i>Allium cepa</i> to 0.01, 0.1 and 1 g/L of nano PS sized 50 nm for 72 hours	Cytotoxicity was induced at the lowest dose of nano PS; nano PS was taken up and accumulated in different organelles.	Accumulated MPs could be transferred to different trophic levels.	[39]

Plant	Condition for MPs phytoremediation	Observation of MPs phytoremediation	Implication on phytoremediation	Reference
Mangrove habitat at Maowei Sea, China	Analysis of abundance, distribution and characteristics of MPs	MPs distribution differed in rhizosphere and non-rhizosphere; colorful MPs were more abundant in the rhizosphere.	The reasons for the differences in MPs distribution in rhizosphere and non-rhizosphere could not be ascertained; the implications of variable distribution on MPs phytoremediation were not clear.	[38]
<i>Betula pendula</i> Roth,	Exposure of potted <i>Betula pendula</i> to fluorescent PA MP beads with a size range of 5 – 50 $\mu\text{m}$	MPs entered the roots and were incorporated into young roots; MPs < 10 $\mu\text{m}$ in diameter were more abundant in the roots, indicating MPs size could be a limiting factor.	The uptake and accumulation of MPs in roots may contribute to rhizofiltration but the actions are localized; there is limited evidence of MPs degradation in roots.	[48]
<i>Lepidium sativum</i>	Exposure of <i>Lepidium sativum</i> to nanoplastics and MPs sized <100 nm and < 5 mm respectively for 72 hours	Germination of seeds exposed to nanoplastics and MPs was retarded due probably to blocked pores on seed capsules. MPs were observed to accumulate on root hairs.	MPs may have deleterious effects on plants; MPs adsorption on root hairs without translocation may be desirable.	[41]
<i>Arabidopsis thaliana</i>	Exposure of <i>Arabidopsis thaliana</i> to functionalized PS nanoplastics, i.e. PS-NH <sub>2</sub> (positively charged) and PS-COOH/ PS-SO <sub>3</sub> H (negatively charged)	Functionalized PS, regardless of the charge, accumulated in <i>Arabidopsis thaliana</i> . Though root exudates promoted aggregation of PS on roots, it reduced the uptake of PS-NH <sub>2</sub> , causing lower accumulation in the root tips; surface charge affects the accumulation of nanoplastics in plants.	Root exudates could promote rhizostabilization and reduce rhizofiltration depending on the surface charges of MPs; the possibility of MPs desorption and transfer along the food chain cannot be ruled out.	[40]
An urban tidal wetland	Analysis of the abundance and distribution of MPs	Vegetation-free mudflats contained the most microfibers while vegetated channel edges had the most microfragments; MPs accumulation in vegetated wetlands was significant and their spatial distribution was affected by hydrodynamics and vegetation.	Wetlands serve as an interceptor and a source of MPs.	[42]
<i>Phragmites australis</i>	Co-exposure of <i>Phragmites australis</i> to PE MP and Cu or Cd	Experimental plants accumulated 70 times more Cu or Cd than the control plants and the presence of PE MPs did not interfere with the efficiency of phytoremediation.	Co-exposure to MPs may have limited impacts on Cu and Cd phytoremediation by <i>Phragmites australis</i> .	[49]
<i>Solanum photeinocarpum</i> , <i>Lantana camara</i>	Co-exposure of the plants to PE MPs, Cd, Pb and Zn	Shoot phytoextraction of Cd was substantially reduced, while rhizofiltration of Zn by <i>S. photeinocarpum</i> increased significantly; at 0.1%, PE MPs lowered shoot phytoextraction of Pb and Zn; at 0.5% and 1% PE MPs, Pb rhizofiltration and shoot phytoextraction of Zn increased.	PE MPs may have variable effects on phytoremediation depending on their concentrations; generally, they adversely affected Cd and Pb phytoremediation.	[50]

While plants could phytoextract, rhizofilter, and rhizostabilize plastic particles, the presence of MPs in the environment could affect the phytoremediation of other pollutants. Variable results on the effects of MPs on phytoremediation were obtained. Upon co-exposing

*Phragmites australis* (common reed) to PE MPs and Cu or Cd, PE MPs did not seem to interfere with the ability of the plant to remove Cu or Cd [49]. Contrarily, for *Solanum photeinocarpum* (American black nightshade), exposure to MPs caused reduced Cd phytoextraction by the shoot and enhanced Zn rhizofiltration of the plant. The concentrations of PE MPs also affected phytoremediation, where low PE MPs concentrations (0.1%) caused decreased shoot phytoextraction of Pb and Zn [50].

### 3.2. Practicality of MPs phytoremediation.

In comparison to other methods of MP removal, especially bioremediation and biocatalysis, phytoremediation to remove MPs has not gained equivalent popularity. The reasons could be attributed to the inherent limitations of applying phytoremediation to MPs. While plants have been shown to rhizostabilize MPs or nanoplastics by facilitating their aggregation, adsorption, and retention on roots or in the rhizosphere (Figure 1), it was also reported that the adsorption was mainly weak and desorption from roots was likely (Table 1). MPs retained in the rhizosphere could also be transported by the soil biota to different parts of the soil via their movements [51]. Similarly, MPs deposited on leaves may be resuspended through wind action or washed off by the rain, thus entering soil and water bodies [43] (Figure 1). Phytoremediation may have localized effects on MPs pollution, leading to a reduction in the MPs abundance of the rhizosphere or zones of phytoremediation, but MPs pollution is a widespread problem that extends beyond these zones to all compartments of the environment. This means that extensive or global phytoremediation needs to be carried out on areas contaminated by MPs. Even so, it may not solve the MPs problem since MPs are found in the air, marine ecosystems, and remote regions that may not be suitable for phytoremediation. As such, vegetation is perceived as a temporary sink by multiple researchers, from which MPs could be released into the environment, thus also making them a source of MPs [35, 42, 43]. It was also pointed out that the adsorption of MPs by plants could hit a saturation point, and there are various environmental conditions that may alter MPs adsorption and promote their desorption [35]. There is a lack of evidence on the effectiveness of rhizostabilizing MPs.

Furthermore, no studies have indicated the degradation of MPs upon phytoremediation. This means that MPs gather on or are taken up and accumulated by plants without being degraded, probably due to a lack of metabolic pathways in plants to utilize the carbon in MPs, unlike bacteria, fungi, and algae. Without being degraded, MPs remain in or on plants and could be transferred to other trophic levels of the food chain when animals feed on them [39] (Figure 1). Studies have demonstrated that plants could take up plastic particles in a size-dependent manner, which implies that phytoextraction and rhizofiltration may help with MPs removal [39]. [48]. No evidence has pointed to the biodegradation of MPs upon entering plants. With the translocation of MPs from the roots to other parts of the plant, the likelihood of MPs transfer to other trophic levels of the food chain increases, resulting in the exposure of organisms feeding on the plants to MPs. This gives rise to ecological risks and concerns about MPs ecotoxicities. MPs may also have toxic effects on plants, albeit low and uncertain [41, 52]. In addition, the presence of MPs could compromise the effectiveness of plants in carrying out phytoremediation, for instance, to remove heavy metals [50]. Nonetheless, more studies are required to ascertain the effects of MPs on the phytoremediation of other pollutants since it is very likely for MPs to co-exist with other pollutants in view of their environmental prevalence [53]. At present, the use of phytoremediation to alleviate MPs pollution may lack



practicality and feasibility as MPs pollution is universal rather than localized, and plants lack the metabolic processes to utilize the carbon in MPs. This lack of feasibility may be partly overcome by combining phytoremediation with bioremediation. While no studies have currently looked at the combined effect of bioremediation and phytoremediation on MP degradation, soil bacteria of the *Pseudomonas*, *Escherichia*, *Bacillus*, *Thermobifidia*, and *Thermomonospora* genera have demonstrated the ability to degrade plastics [54]. In addition, soil also contains plastic-degrading fungi such as those of the phyla Ascomycota, Basidiomycota, and Mucoromycota [55].

#### 4. Conclusions

Plants have been shown to interact with MPs via adsorption, aggregation, uptake, and translocation, leading to the potentiality of phytoextraction, rhizofiltration, and rhizostabilization of MPs. However, MPs do not seem to be degraded by plants, and translocation of MPs in plants could occur, causing MPs to appear in other parts of the plants in addition to the roots. Ingestion of the plant parts could transfer MPs from plants to feeding organisms and other consumers in the food chain. It is likely that MPs intercepted by vegetation are released, making plants a temporary sink and a source of MPs. MPs can negatively affect the growth and health of plants while altering their ability to remove other pollutants. In view of a lack of evidence on the effectiveness of phytoremediation of MPs and the associated drawbacks, phytoremediation in its current form may not be practical for MPs removal. To enhance practicality, bioremediation through stimulation or augmentation of the microbial community in the rhizosphere for the biodegradation of MPs is more feasible. Plants can play an auxiliary role in the bioremediation of MPs-contaminated soil by providing anchorage for plastics-degrading microorganisms or stimulating them, for instance through root exudates.

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#### Competing Interest

The author declares that there is no competing interest.

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