

Review

# Microplastics and Potentially Toxic Elements: Potential Human Exposure Pathways through Agricultural Lands and Policy Based Countermeasures

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**Citation:** Igalavithana, A.D.; Mahagama, M.G.Y.L.; Gajanayake, P.; Abeynayaka, A.; Gamaralalage, P.J.D.; Ohgaki, M.; Takenaka, M.; Fukai, T.; Itsubo, N. Microplastics and Potentially Toxic Elements: Potential Human Exposure Pathways through Agricultural Lands and Policy Based Countermeasures. *Microplastics* **2022**, *1*, 102–120. <https://doi.org/10.3390/microplastics1010007>

Academic Editor: Nicolas Kalogerakis

Received: 17 December 2021

Accepted: 19 January 2022

Published: 21 January 2022

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**Abstract:** Microplastics (MPs) have been identified as an emerging soil pollutant and a global environmental concern. Scientists have recently paid attention to the contamination of soil by MPs as their detrimental impacts on soil systems are largely unknown. MPs are considered to be vectors for other soil contaminants, such as potentially toxic elements (PTEs) and organic contaminants. PTEs are persistent contaminants and are often released into soils in large quantities. MPs adsorb PTEs, mainly via electrostatic attraction and surface complexation, and increase their mobility in soils. These complexes can be easily absorbed by plants; hence, the accumulation of PTEs in plants can be enhanced in both microplastic and PTE contaminated soils. Furthermore, there is a high risk of food chains contamination by PTEs due to crops grown in both microplastic and PTE-contaminated soils. Consequently, countermeasures including policy- and governance-based approaches that target circular economy as well as reduce, reuse, recycle (3R) applications are being discussed around the world to minimize the environmental contamination of MPs.

**Keywords:** microplastics; toxins; plastic-related toxins; plant absorption; contamination path-way; policy and governance

## 1. Introduction

Plastic production and utilization have increased tremendously during the past couple of decades and reached a remarkable level with the COVID-19 pandemic [1]. In the COVID-19 pandemic 1.6 million tons/day of plastic waste has been generated globally, mainly due to the increased production of disposable personal protective equipment; this means there was an annual generation of 75 kg of plastic waste per capita [2,3]. These anthropogenic synthetic materials have accumulated in soils due to several reasons, such as improper waste disposal, the application of sewage sludge as fertilizers, soil mulching,

and precipitation [4–7]. It has been identified that more than 95% of the microplastics (MPs) in domestic wastewater are transferred into sewage sludge at wastewater treatment plants [8]. The application of this contaminated sewage sludge to agricultural lands adds a substantial amount of MPs to those lands [9]. Furthermore, plastic-coated fertilizers are another direct intestinal application of MPs into agricultural lands that has occurred over the last two decades, and now report accumulations and consequent effects [10]. Mulching plastic sheets annually add a large amount of MPs to agricultural soils, which showed a significant impact on soil bulk density and water infiltration. Furthermore, those MPs tend to accumulate in soil while increasing the absorption by plants and alter biological functions [11,12]. Moreover, recent studies have very clearly shown that there is a higher accumulation of MPs in soils than in aquatic environments, and a potential entering of MPs in the terrestrial food web [13,14]. However, soil pollution by MPs has received insufficient attention compared to marine ecosystems [13].

Microplastics can cause adverse impacts on the biodiversity and quality of soils due to their high resistance to biodegradation [15]. MPs tend to be directly ingested by soil macroorganisms and accumulate in body tissues, unfavorably affecting their survival. Furthermore, the accumulation of MPs in soils might negatively impact beneficial microbial populations in soils [16].

In addition, MPs in soils act as adsorbents for both organic and inorganic contaminants [17–19]. Potentially toxic elements (PTEs) are important inorganic soil contaminants that cause negative impacts on soil and food quality [20,21]. PTEs are easily absorbed by plants; hence, the production of foods in contaminated agricultural soils threatens the quality of said foods [20]. PTEs in the soils can be adsorbed onto the surface of MPs, which are simply absorbed by plants. Moreover, PTEs adsorbed onto MPs have shown high mobility in soils due to their low weight and size; hence, surrounding soils can be contaminated with both MPs and PTEs [22,23]. Weathered MPs in the environment have shown a higher capacity to adsorb PTEs than their pristine forms [24,25].

The studies conducted on MPs and their potential to be vectors for PTEs in soils have received minor attention. Hence, this review has the focus of discussing the (1) mechanisms of the adsorption of PTEs onto MPs and their transportation in soils, (2) environmental factors affecting the adsorption of PTEs onto MPs, (3) plant adsorption of PTE-adsorbed MPs and their toxicity, and (4) research gaps, future prospects, and possible countermeasures, focusing in particular on policy and governance.

## 2. Adsorption of Potentially Toxic Elements onto MPs

MPs serve as vectors for both potentially toxic elements (PTEs) (i.e., Fe, Mn, Al, Pb, Cu, Ag, and Zn) and hydrophobic organic contaminants (i.e., persistent organic pollutants (POPs) such as polychlorinated biphenyls (PCBs), organochlorine pesticides (OCPs), and polyaromatic hydrocarbons (PAHs)) [26]. The adsorption of PTEs onto MPs and their possible transportation mechanism via MPs are discussed in this section.

### 2.1. Adsorption Mechanisms of Potentially Toxic Elements on MPs

It has been reported that the adsorption of PTEs onto MPs occurred at high concentrations [26]. Holmes et al. [27] reported PTEs adsorbed on polyethylene resin pellets collected from the beaches in Southwest England, and Mohsen et al. [28] observed adsorbed PTEs (i.e., As, Cd, Cr, Cu, Mn, Ni, Pb, and Zn) on isolated MPs from sea cucumber culture ponds in China. In a study conducted to evaluate the PTEs adsorption on MPs of chlorinated polyethylene (CPE), Polyvinyl chloride (PVC), and two types of polyethylene plastic particles of LPE and HPE showed the high adsorption capacities of  $Pb^{2+}$ ,  $Cu^{2+}$ , and  $Cd^{2+}$  on all MPs [19]. The  $Pb^{2+}$  showed strong adsorption compared to the others due to strong electrostatic interaction. Moreover, the considered PTEs showed the highest sorption affinity to chlorinated polyethylene due to its high polarity and negativity created with the

presence of chlorine [29]. The chemical structure and electronegativity of MPs are responsible for the adsorption of PTEs. Differently, the adsorption process seems to not be affected by the crystallinity [19]. Gasperi et al. [30] investigated airborne fibrous MPs and transport pollutants through MPs, further mentioning that airborne fibrous MPs may carry pollutants adsorbed from the surrounding environment due to their hydrophobic surface. According to the study by Holmes et al. [27] metal adsorption may have occurred through the interactions between bivalent cations (e.g.,  $\text{Cu}^{2+}$ ,  $\text{Cd}^{2+}$ , and  $\text{Pb}^{2+}$ ) and oxyanions (e.g.,  $\text{Cr}_2\text{O}_4^{2-}$ ) with charged or polar regions of the plastic surface and via non-specific interactions between neutral metal–organic complexes and the hydrophobic surface of the bulk plastic medium. Furthermore, the adsorption capacity of MPs can be varied depending on their morphology, specific surface, or the presence of additives [17].

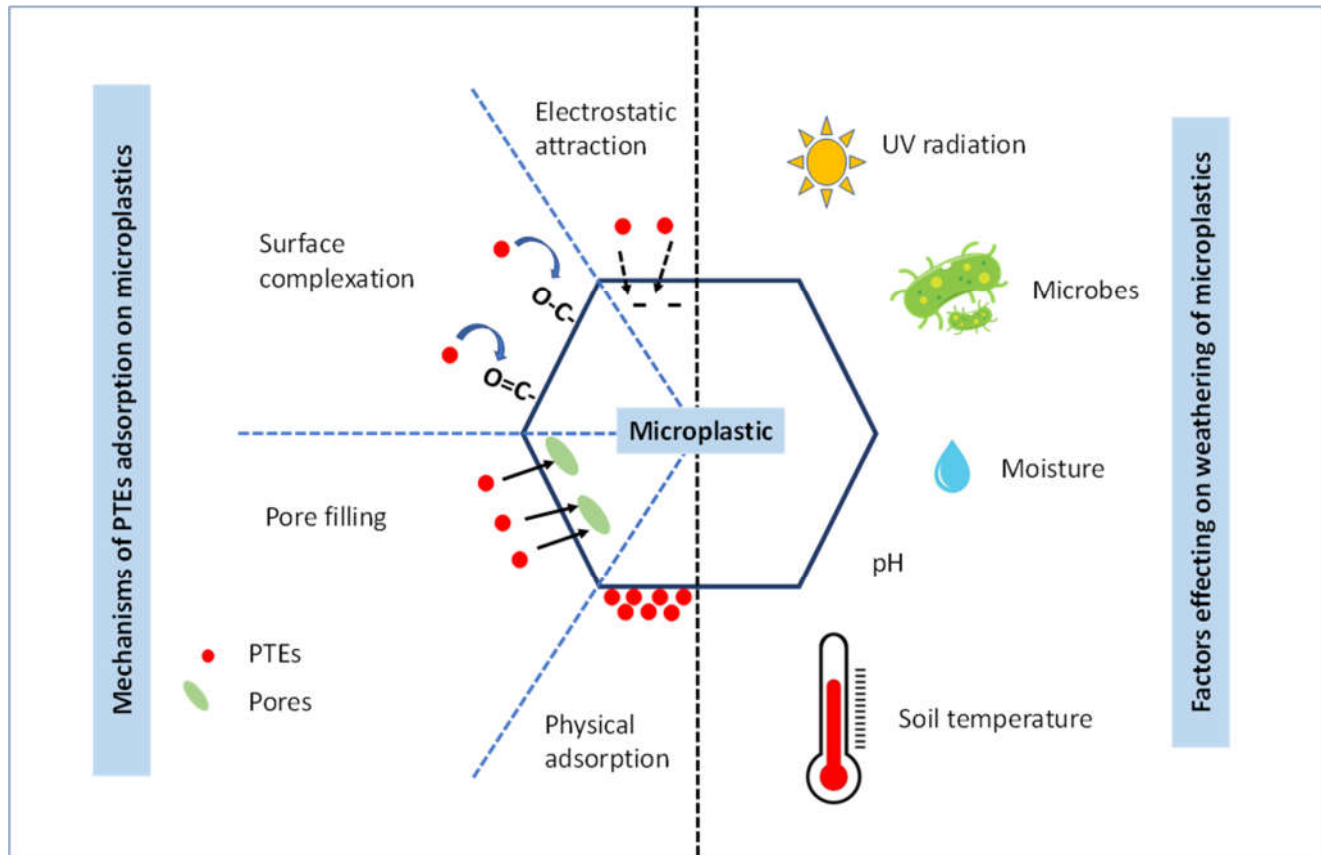
Physisorption has been identified as an important mechanism of PTEs adsorption on MPs with simple structure and polymer types and MPs with low density of surface functional groups [9,31]. However, PTEs adsorption on MPs mainly occurs via chemisorption mechanism, hence, the surface area and the functional groups are highly important. Moreover, electrostatic attraction of positively charged PTEs on negatively charged MPs has been identified (Figure 1) [19,32]. For instance, in a study conducted by Zou et al. [19], the adsorption of  $\text{Pb}^{2+}$  onto MPs was mainly governed by electrostatic attraction and that of  $\text{Cu}^{2+}$  and  $\text{Cd}^{2+}$  was due to both electrostatic attraction and surface complexation. Similarly, Wang et al. [33] studied the  $\text{Cu}^{2+}$  adsorption by polyethylene MPs and observed that the highest adsorption (0.911 mg/g) occurred due to the electrostatic interaction. Furthermore,  $\text{Pb}^{2+}$  showed the highest affinity towards the MPs due to the lowest hydrated ionic radius compared to the  $\text{Cu}^{2+}$  and  $\text{Cd}^{2+}$  [19]. The surface complexation of PTEs was supported by the functional groups  $\text{C}-\text{O}/\text{C}=\text{O}$  [19]. Moreover, the fourier-transform infrared spectroscopy (FTIR) and X-ray photoelectron spectroscopy (XPS) C 1 s spectra for polyamide, polyvinyl chloride, polystyrene, acrylonitrile butadiene styrene, and polyethylene terephthalate showed increased  $-\text{C}=\text{O}$  functional groups on MPs (except polyvinyl chloride) and a ratio of  $-\text{C}=\text{C}/-\text{C}-\text{C}$  in polyamide and acrylonitrile butadiene styrene after the adsorption of  $\text{Cd}^{2+}$ . Hence, it was clear that the oxygen containing functional groups and cation– $\pi$  electron interaction were vital in  $\text{Cd}^{2+}$  adsorption on MPs [34]. In addition, Shen et al. [35] revealed that addition of surfactants increased the hydrophilicity and negative charge of MPs surface, and therefore, increased the adsorption of  $\text{Pb}^{2+}$  on MPs. The cationic PTEs sorption also relied on the solution pH. According to a study conducted by Zhou et al. [34], the  $\text{Cd}^{2+}$  adsorption on the MPs was increased then decreased in a range of pH from 6.0 to 9.0. The increased surface negativity which facilitated the electrostatic attraction among MP and  $\text{Cd}^{2+}$  might be the reason for high  $\text{Cd}^{2+}$  adsorption at low pH values [36]. The adsorption of PTEs onto MPs has been described as two steps according to the Weber–Morris model. First, PTEs are rapidly adsorbed onto active sites of MPs via covalent and van der Waals forces; when the adsorption sites are saturated, PTEs are diffused onto the pores of MPs [37].

Scientists also studied the anion adsorption by MPs. For instance, Dong et al. (2020) used Gaussian 16 to simplify the polystyrene structure and revealed that the surface of polystyrene contained positive electrostatic potential (0–14 kcal/mol). Mainly, the positive electrostatic potential was found in H atoms located on hydroxyl (i.e., +46.96 kcal/mol) and carboxyl groups (i.e., +56.60 kcal/mol), and it facilitated the adsorption of oxyanions of arsenic [38].

## 2.2. Transport of Potentially Toxic Elements via MPs

The sources of PTEs in the environment are diverse, including natural weathering and mining, industrial effluents, urban runoff, agricultural activities, and many others [39,40]. The most studies have concerned a high load of PTEs on the surface of MPs, documenting the potentiality of MPs as an alternative source or sink of PTEs contaminant in the environment [26]. As discussed above, MPs may correspond to a habitat and transport

medium for PTEs. As vectors for PTEs, MPs in particular speed up their transport in diverse environments. Due to the small sizes of MPs, they can easily transfer through sediments, sands, and soils [41], and the micro pores or tunnels formed by plant roots or animal activities may enable the transport of large particles of MPs through the soil column [42].



**Figure 1.** Adsorption mechanisms of potentially toxic elements (PTEs) onto microplastics (MPs). UV: ultraviolet.

The association of PTEs with MPs in aquatic environments primarily depends on the surface properties of MPs and the environmental conditions [43]. Furthermore, the transportation of MPs and attached PTEs in aquatic systems largely depends on the water flow [44]. Therefore, the movement of MPs through the water flow and the adsorption of PTEs on MPs can easily transport the PTEs in the environment. In addition, the PTEs are used as additives in plastics; hence, there is a risk in the releasing of those to the environment. For instance, Garçon et al. [45] observed PTEs leached from 15 different types of disposable protective gloves irrespective of the colors and materials of them. Soaking of gloves for 40 h in 0.4 M  $\text{HNO}_3$  + 0.05 M HF released 60 different types of elements and most of them were belonging to PTEs (i.e., Fe, Hf, Mg, Mn, Pb, Rb, Sr, Sn, Ti, Zr, Ag, As, Ba, Ce, Cr, Cu, Ga, La, Li, Nd, Ni, Sc, Se, Th, Tl, V, and Y). Furthermore, a study conducted by Turner et al. [46] reported bioaccessible concentrations in added Pb at a range of 20–1200  $\mu\text{g g}^{-1}$  in MPs, and it was around  $10^2$ – $10^4$  times higher than that in adsorbed Pb. In addition, different organic additives (e.g., Phthalates, Bisphenol A, Poly-brominated flame retardants, and Nonylphenol) can be released from the MPs, and those also could act as vectors of PTEs [47,48].

Isobe et al. [49] documented that the size and density of MPs also determine the sedimentation and transport in water sources. Furthermore, MPs which had a high density

that was greater than the water slowly settled into the sediments. However, special characteristics of MPs affect the sorption/desorption of PTEs and their movement in the aquatic environment and sediments. The relationship of PTEs with MPs seems to be more of a general mechanism concerning different types and sizes of MPs. Thus, MPs might be considered as a vector for the transport of PTEs in aquatic systems [27]. Ultimately, these PTEs either bioaccumulate in food chains or are released back into the environment in a biologically available soluble form. Abbasi et al. [22] documented that PTEs transported by MPs were not native to the plastic but derived from the environment, indicating that the PTEs on the MPs were accumulated from surface sediments; they found that MP particles can act as a carrier and transport PTEs to the rhizosphere zone. However, very limited research has concentrated on the interaction between MPs and PTEs.

In recent years the atmospheric transport of MPs has been considered to be an important vector that could accelerate the deposition of MPs onto land or aquatic environments [50]. Atmospheric MPs vary greatly in their shapes; these densities and shapes of MPs particles are important properties in regard to their transportation activities [51]. Moreover, Wang et al. [44] documented the fast sorption/desorption kinetics of airborne contaminants on small-size MPs particles, which are important to adsorb pollutants during the air movement, especially at the cross-boundary movement of MPs. However, the interactions between MPs with PTEs in the atmosphere and their impacts on the environment, humans, and ecosystems are unstudied [52].

### 3. The Effect of Environmental Factors on the Adsorption of PTEs onto MPs

As discussed above, MPs can be carriers of PTEs. Plastic waste in the environment can be weathered and aged due to physical, chemical, and biological factors [11]. Research has explained that many biodegradable plastics are susceptible to disintegration rather than degradation [53]. These processes create MP particles which are <5 mm; further degradation and disintegration of MPs can generate nanoplastics which are <0.1  $\mu\text{m}$ . These micro- and nanoplastics also undergo various weathering processes in the environment due to factors such as ultraviolet radiation, microbial degradation, physical disintegration, and chemical oxidation [54,55], which enhance the oxygen-containing functional groups, hydrophilicity, and surface charge of MPs. Consequently, environmental weathering increases the PTEs adsorption capacity of MPs [37,55,56].

Ultraviolet-radiation-induced weathering is considered to be an important step of the environmental consequences of MPs. MPs absorb ultraviolet radiation and create polymer radicals, followed by surface oxidation. Dong et al. [57] also observed increased surface oxygen content after the environmental weathering of MPs; they suggested that the O/C ratio is a reliable indicator of the oxidation degree of MPs. Furthermore, due to this ultraviolet-radiation-induced weathering, polymer chains can be cleaved and form ester, aldehyde, formate, and propyl surface functional groups, which facilitate the adsorption of PTEs [24,25]. MPs in the soils can also be subjected to biodegradation, which occurs due to the biochemical transformations mediated by microorganisms. The biodegradation of MPs is facilitated by ultraviolet-radiation-induced weathering, abiotic hydrolysis, and physical disintegration. Those processes increase the surface area and decrease the molecular weight of MPs, which ultimately increase the microbial colonization [25,58]. Environmental conditions such as moisture, temperature, oxygen, and suitable population of microorganisms affect the biodegradation of MPs in soil [59]. For instance, degradation of MPs is enhanced in high moisture contents and high temperatures of soils. In addition, these conditions can also accelerate the ultraviolet-radiation-induced weathering of MPs [25]. However, the MPs weathering and aging under natural conditions are complicated as several agents expose together on MPs particles [60]. Weathered MPs have been shown to have a high adsorption capacity of PTEs due to the high number of surface functional groups, surface area, microcracks, surface roughness, crystallinity, polarity, functional groups, carbonyl index, leachates, and sorption capacity [38,57,60]. Furthermore, it has been identified that the adsorption of PTEs increases with the amount of biofilm generated

on the surface of MPs [18]. Biofilm formation alter the surface properties of MPs which lead to increase PTEs adsorption via increased wettability, and functional groups due to extracellular polymeric substances of microorganisms [61].

In addition, dissolved organic matter has been identified as a major factor that could increase the adsorption of PTEs onto MPs [17]. Dissolved organic matter increases the adsorption of PTEs onto MPs in two ways. First, PTEs adsorb onto dissolved organic matter, then those make complexes on the surface of MPs. Second, dissolved organic matter constructs a complex with MPs and then adsorbs PTEs [17,62]. Moreover, PTEs can be adsorbed onto the organic matter due to the high density of the surface functional groups [61,63,64]. The adsorption of PTEs onto MPs also depends on the pH, the presence of ions in the solution [64,65]. The pH of the solution affects the surface charge of MPs and the speciation of PTEs; hence, a prevailing suitable pH value in the soil will increase the adsorption of PTEs onto MPs. For instance, Li et al. [64] first observed an increase followed by a decrease in the adsorption of Cd onto microplastics within the pH range of 5–9 as discussed in Section 2.1. Moreover, authors have observed decreased Cd adsorption onto MPs with increased ion concentrations. Hence, environmental factors play a major role in the adsorption of PTEs onto MPs in the soil.

#### 4. Effects of MPs on the Bioaccumulation and Toxicity of PTEs

During the last hundred years, population growth and industrialization have grown at a fast rate and increased the demand for the exploitation of the Earth's natural resources and synthetic materials like plastics [66,67]. Currently, the environment has been seriously polluted by both PTEs and plastics due to those anthropogenic activities [68,69]. Due to the uncontrolled discharge of waste and wastewater containing PTEs from various sources, such as industries, mines, agricultural activities, vehicles, batteries, and PTE-containing paints, their concentration is increased in soils around the world [70,71]. Of particular note is that PTEs are non-biodegradable, considered toxic or carcinogenic ions, tend to accumulate in organisms [72–74], and may destroy enzyme activity, leading to impaired biological growth and metabolism as well as oxidative damage to living beings [75,76]. Moreover, PTEs can bioaccumulate along the food chain and ultimately threaten human health [77,78]. At present, research on the combined effects of MPs and PTEs, draws broad attention since MPs act as carriers for PTEs [32,79–82]. In addition, additives in plastics (e.g., pigments, plasticizers, PTEs) have shown potential to leach from aged MPs, releasing potentially hazardous chemicals including PTEs to the soil [83].

##### 4.1. Microplastic Uptake by Plants

MPs are abundant throughout the environment and cause considerable concern because micro- and nanoparticles are small enough to be absorbed by many plants in addition to the fact that their potential for bioaccumulation is hazardous because they can permeate biological membranes [84,85]. Agricultural lands in particular are highly vulnerable to MPs due to the application of organic fertilizer coming from municipal solid waste and sewage sludge, organic manures, irrigation, and runoff water, as well as due to intensive agricultural practices like plastic mulching [86]. Furthermore, the MPs are prone to transport processes in soil environment, depending on both physicochemical properties of soils and plastic grain size [87]. In addition, soil macroorganisms are playing an important role in the movement of MPs in soil vertically and into groundwater via casts, burrows, egestion, and adherence organisms' exterior [60]. Therefore, with respect to the human health concern, the absorption mechanism and accumulation process of MPs in individual plants have been studied [88,89]. As shown in Table 1, rice, maize, onion, beans, carrot, corn, oat, lettuce, wheat-like commercial agricultural plants, and other plant species such as grasses and aquatic plants were highly vulnerable to exposed MPs under current anthropogenic activities. Importantly, it was found that MP material was stacked and adsorbed onto the root surface in addition to having entered the shoots of plants through the root system along with water and nutrients, using the transpiration pull of

the vascular system [88–90]. The sizes, shapes, and chemical properties of microplastic particles directly affect their translocation in plants [88]. Importantly, MPs are highly potential and are likely to be absorbed by plants' well-developed root systems [89,90]. Furthermore, plants' root growth, movement, and water absorption process greatly influence the migration of MPs [89,91,92]. Based on the above literature, it can be concluded that MPs can enter a plant through the free space between root cells and require a certain transpiration pulling force to reach the aboveground part of the plant. Additionally, MPs might accumulate in the edible and non-edible parts of the plants [93]. However, studies are still lacking on the distribution of microplastics through food chains.

**Table 1.** MPs reported in agricultural plants and other plant species.

Microplastic/Plastic			Plant	Effect	Reference
Types	Size (mm)	Concentration (% w/w)			
Polystyrene (PS)	0.55~56	2	Onion ( <i>Allium fistulosum</i> )	Increased root biomass, total root length, and mean diameter	[50]
	<0.00005	-	Garden onion ( <i>Allium cepa</i> )	The root length was inhibited	[94]
	0.1–0.15	0.1–10	Corn ( <i>Zea mays</i> )	Plant root biomass decreased	[44]
	D = 0.001	-	Garden lettuce ( <i>Lactuca sativa</i> )	The PS is transported through the vascular system to the stem and leaves	[95]
	0.0001	5	Wheat ( <i>Triticum aestivum</i> )	The root length increased, the root/shoot ratio decreased, and the biomass increased	[96]
	5	-	Broad bean ( <i>Vicia faba</i> )	Decreased biomass and catalase enzyme activity in addition to blocking cell connections or cell wall pores for the transport of nutrients in roots	[89]
	D = <0.001	-	Rice ( <i>Oryza sativa</i> )	PS was mostly aggregated in the vascular systems of the roots, stems, and leaves, with a high possibility of entering the food chain	[97]
	<50	-	Rice ( <i>Oryza sativa</i> )	Higher doses of PS caused a ≈40% decrease in shoot biomass	[98]
	0.01	-	Rice ( <i>Oryza sativa</i> )	Affected the transpiration and stomata of rice seedlings primarily via inhibiting their root vigor	[99]
	<0.048	-	Garden cress ( <i>Lepidium sativum</i> )	Significantly declined germination rate and inhibited plant growth	[88]
<0.001	-	Carrot ( <i>Daucus carota</i> L.)	Entered the roots and accumulated in the intercellular layer; particles were able to translocate to the leaves	[90]	
0.001	-	Lettuce ( <i>Lactuca sativa</i> L., <i>Rosa</i> )	Adherence, uptake, accumulation, and translocation of PS in the vascular tissue	[100]	
Low density polyethylene (LDPE)	L: 4–10	1	Garden lettuce ( <i>Lactuca sativa</i> )	The total biomass decreased and the composition of the rhizosphere bacterial community changed	[95]
	L = 6.9; W = 6.1	1	Wheat ( <i>Triticum aestivum</i> )	The fruit biomass and leaf number decreased	[101]

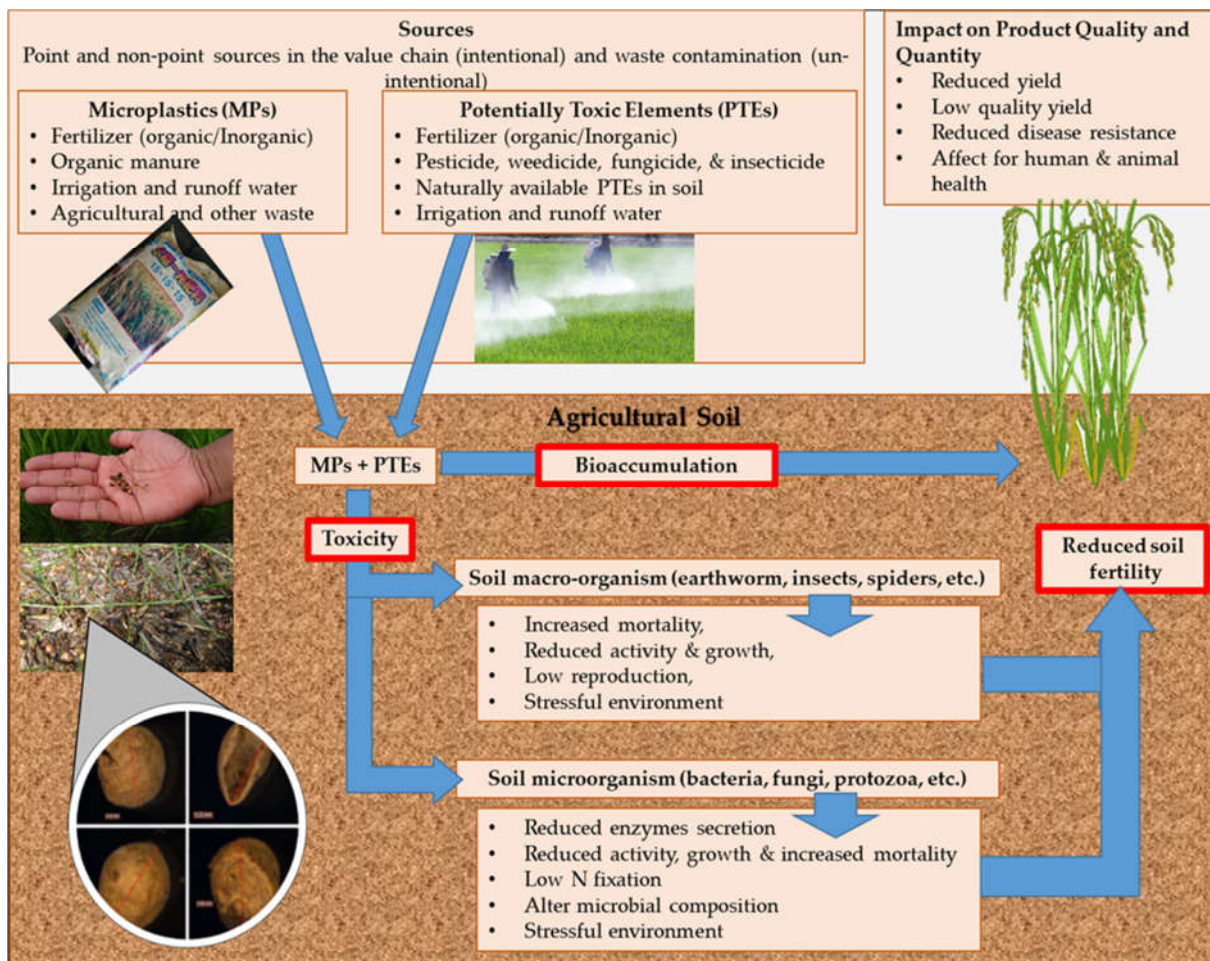
	-	1	Wheat ( <i>Triticum aestivum</i> )	Affected vegetative and reproductive growth	[102]
	0.053–1	0.2–2.5	Bean ( <i>Phaseolus vulgaris</i> )	Aboveground and root biomass affected but the effect was not significant	[91]
	L = 5, W = 5	0.1–0.4	Carrot ( <i>Daucus carota</i> )	Aboveground biomass and root mass decreased with increasing concentration	[103]
Poly lactic acid (PLA)	0.1–0.15	0.1–10	Corn ( <i>Zea mays</i> )	High concentration of PLA significantly reduced plant biomass	[15]
	0.065	0.1–0.001	Perennial ryegrass ( <i>Lolium perenne</i> )	Reduced shoot height and biomass	[16]
	-	-	Bean ( <i>Phaseolus vulgaris</i> )	Root and aboveground biomass reduced	[91]
Polyethylene (PE)	D = 0.2–0.25	0.5–8.0	Wheat ( <i>Triticum aestivum</i> L.)	A high concentration of PE damaged the antioxidant system in wheat roots	[104]
	0.003	-	Corn ( <i>Zea mays</i> )	PE reduced or blocked water and nutrient uptake as well as the growth of the maize plant	[105]
	0.5	0.1–10	Lettuce ( <i>Lactuca sativa</i> L.)	Increased the toxicity, uptake, accumulation, and bioavailability of heavy metals	[93]
High density polyethylene (HDPE)	-	0.1–0.001	Carrot ( <i>Daucus carota</i> )	Shoot height and biomass reduced, fewer seeds germinated	[16]
	0.01–0.15	0.1–10		There was no significant change in plant biomass	[103]
Polyamide (PA)	0.015–0.02	2	Onion ( <i>Allium fistulosum</i> )	Significantly affected plant biomass, root traits, tissue elemental composition, and soil microbial activity	[50]
	0.015–0.02	2	Wheat ( <i>Triticum aestivum</i> )	The total biomass increased as did the total root length and mean diameter	[96]
Polypropylene (PP)	-	0.02	Garden cress ( <i>Lepidium sativum</i> )	Occurrence of oxidative burst	[106]
	L = 5, W = 5	0.1–0.4	Carrot ( <i>Daucus carota</i> )	Aboveground biomass and root mass decreased with increasing concentration	[103]
Polyester fibers (PFs)	L = 5, D = 0.008	0.2	Onion ( <i>Allium fistulosum</i> )	Significantly changed plant biomass, root traits, tissue elemental composition, and soil microbial activity	[50]
	L = 1.3, D = 0.03	-	Grasses ( <i>Festuca brevipila</i> ) and herbs ( <i>Achillea millefolium</i> )	Decreased biomass	[107]
Polyether sulfone (PES)	L = 5, D = 0.008	0.2	Onion ( <i>Allium fistulosum</i> )	The total biomass and root biomass increased, the total root length and mean diameter increased, and the root microbial activity increased	[50]
	L = 1.3, D = 0.03	0.4	Wood small-reed ( <i>Calamagrostis epigejos</i> )	The root biomass increased	[107]
Expandable polystyrene (EPS)	8.3	-	Mung bean ( <i>Phaseolus radiates</i> ), lettuce	Low levels of interaction with the crop dependent and water absorption rate	[92]



				( <i>Lactuca sativa</i> ), and rice ( <i>Oryza sativa</i> )		
Polyvinyl chloride (PVC)	0.018–0.15	0.5–2	Lettuce ( <i>Lactuca sativa</i> L.)	PCV-a promoted carotenoid synthesis whereas PVC-b inhibited it	[95]	
Melamine phenolic (MP)	0.0048	-	Garden cress ( <i>Lepidium sativum</i> )	Accumulated on the root hairs, the germination rate was significantly reduced, and pores in the seed capsule were physically blocked	[88]	
Polyetherimide (PEIs)	-	0.01–0.1	Oat ( <i>Avena sativa</i> ) and radish ( <i>Raphanus sativus</i> )	Nitrogen released from the tested PEIs but no harmful effect; harmful to plants only at high concentrations	[108]	

#### 4.2. Effect of MPs on Soil Animals and Microbial Activity

Soil contamination due to microplastics might affect the physical, chemical, and biological properties of soil. This might significantly influence the functional and structural diversity of soil animals (earthworms, insects, and spiders) and microbial communities (bacteria, fungi, and protozoa), causing serious damage to the agricultural soil [50,109,110] (Figure 2). These changes in soil properties and composition directly affect the fertility of the soil by decreasing the growth and activities of soil animals and microbial communities [75,90,111]. The soil is an important boundary where plants act together with beneficial and harmful microbial communities. The pollution of soil by high-density polyethylene (HDPE) decreases soil pH, which might impact microbial growth and functions [16,112]. Interestingly, recent studies have reported an increase in soil pH and a reduction in soil electrical conductivity in polyethylene (PE)-, polylactic acid (PLA)-, and low-density polyethylene (LDPE)-contaminated soils [44,111]. Changes in soil pH are highly influenced by variations in soil microbial communities [104]. Furthermore, a few studies have discovered adverse effects of polystyrene (PS) particles on microbial activities in the soil as well [113,114]. Despite the above, the mechanism of pH change in soil caused by the contamination of MPs is still unclear [115]. Thus, more studies are recommended to understand the changes in pH under the influence of MPs. Interestingly, polypropylene (PP) was found to have a positive impact on bacterial growth in the soil [116]. Wang et al. [44] reported that microplastic particles affect the abundance of arbuscular mycorrhizal fungi in the soil. Additionally, Chen et al. [117] reported that poly lactic acid (PLA) affects microbial-supported mineral absorption and nitrogen fixation rates. In addition, Fei et al. [118] found that polyethylene (PE) and polyvinyl chloride (PVC) encourage acid phosphatase and urease activity in soil. Similarly, Huang et al. [119] exposed the fact that microplastics provide exceptional habitats for some microbial communities, such as pathogens and plastic-degrading bacteria. In addition, studies showed that due to the changes in microbial community composition in soils, the nutrient cycling also can be significantly altered. For instance, Seeley et al. observed polyurethane foam or polylactic acid MPs amended sediments increased nitrification and denitrification, while that of decreased by polyvinyl chloride MPs. Those results imply that the nitrogen cycle in sediments can be significantly impacted by MPs, as MPs are serving as organic carbon substrates for microbial growth [120]. However, the interaction of soil animals and microorganisms with MPs largely remains unexplored. Thus, the limited information available on general microbial activity in soil is influenced by MPs. Therefore, further investigations are strongly recommended to understand the positive and negative effects of microplastics on soil animals' and microorganisms' activities in the soil.



**Figure 2.** Exposure and bioaccumulation of MPs and PTEs in agricultural soil.

#### 4.3. MP Accumulation in Plants and Toxicity

The MPs act as vectors and sinks of toxic organic as well as inorganic pollutants. The accumulation of MPs in plants could pose a potential human health risk. The accumulation and transmission of MPs in the food chain is an important way for them to be exposed to the human body [121,122]. Researchers have reported the possible accumulation of MPs in rice [85], lettuce [93], carrot [90], wheat [102], mung bean [123], and beans [89,91]. Therefore, the risk of bioaccumulation of MPs through the food chain is very high, ultimately threatening human health. Importantly, MPs have strong hydrophobicity and a strong adsorption capacity for heavy metals, organic pollutants, pharmaceuticals, pesticides, herbicides, weedicides, and insecticides, very hazardous substances from agricultural soils. Thus, through food chains, MP particles can enter the human body and might enter human organs through the circulatory system [124]. Currently, only a limited number of studies have focused on the size of MPs as a significant factor for plant toxicity [88,89,102]. Few authors have reported the effect of the shape of MPs on plant toxicity [16,50]. Furthermore, the surface charges of available materials in soil and the chemical configuration of MPs are important factors that govern the uptake and toxicity of MPs in plant species [50,102,125]. Qi et al. [111] reported that biodegradable MPs can harmfully influence biometric parameters in wheat at the vegetative and reproductive phases. In addition, Jiang et al. [89] reported that MPs encouraged ecotoxicity and genotoxicity in plants. Similar type of experimental results were reported by a few authors on the effect of MP toxicity on plant growth and performance of lettuce [126], maize [127], rice [99], and maize [16]

productivity. The MP particles concentrated in the soil are possibly adsorbed to the belowground parts of plants, which might lead to reaching absorbed MPs into the root eaters' body [89]. Importantly, MPs can alter the evapotranspiration of plants and contribute to a more pronounced drought condition [128]. This modification in the metabolic system affected a weakening in the harvest yield. Boots et al. [16] noticed that MPs damagingly influence the growth of above and belowground parts of plants. Zhang et al. [52] reported on the influence of MPs on the absorption of Cd in agricultural soil and mentioned that higher concentrations of MPs caused higher absorption of PTEs in addition to possible risks to plants and animals. Thus, MP deposits and particles serve as vectors for other pollutants. These contaminants, after absorbing and accumulating into plants, might lead to a possible path into the food chain. They might govern the bioaccumulation of MPs and other pollutants, which can lead to major health risks. Based on the above literature, the bioaccumulation of MPs affects plants' agricultural productivity and animals' health. Therefore, intensive management and clear policy are required to manage the pollution of MPs in agricultural soils.

### 5. Policy and Governance Measures

Necessary policies and institutionalized governance measures could be key drivers in creating appropriate barriers to MPs entering humans through food chains [129] and thus, the PTEs associated with MPs as well. Furthermore, the integration of 3R and circular economy approaches while restricting the release, recontamination, and spread of MPs and PTEs into the environment could support the mitigation process. Present plastic waste management policies mainly focus on waste disposal and regulation on land while the policies regarding MPs have largely been focused on the prevention of aquatic MP pollution [130]. International and national agencies working on environmental protection and conservation have intervened in the determination of policies and regulations to protect aquatic environments from the harmful impacts of MPs [131,132]. However, attention is growing on the detrimental impacts of MPs in terrestrial systems and the dialog surrounding mitigative policy measures [130,133]. This would support the measures intended to mitigate the contamination of agricultural lands by MPs and PTEs as well as human exposure to them through food chains that originate in land-based agriculture. As elaborated on in Section 4.1, the major contamination pathways, such as irrigation water, fertilizers, and soil conditioners, including sewage sludge, are to be considered in the countermeasuring policies and institutionalization process. For example, the Basel Convention Plastic Waste Amendments, 2019, for the international movement of plastic waste [134] can be further strengthened to include products related to agriculture, such as soil amendments and compost. Figure 3 illustrates the policy- and governance-based countermeasures for MPs and PTEs that reach humans through ingestion. The California State Legislature enacted two bills that require the quantification of MPs in various media and the development of new management strategies to address MP pollution, targeting the mitigation of human exposure through ingestion [135]. The case of California is a classic example of recent policy developments instituted to tackle the contamination of humans by MPs via ingestion.

As discussed in previous sections, the management of sewage sludge plays an important role in the management of the recontamination of agricultural lands by MP-associated PTEs. In particular, practices such as open dumping, added as a soil improvement and used as a fertilizer for crops, lead to recontamination by MPs and associated PTEs. However, so far, policy measures that directly target the link between MPs and sewage sludge are not reflected in policy around the world [136]. Nevertheless, the land application of sewage sludge is regulated and institutionalized in several European Union (EU) countries, where these management practices indirectly restrict soil recontamination [136]. For example, the sewage sludge ordinance in Germany states that sewage sludge from WWTP will not be permitted for direct use as a fertilizer from 2029. However, many other countries around the world, especially in Asia-Pacific, Africa, and the Americas, use

open landfilling or agricultural land application as a primary method of sewage sludge disposal [137]. Furthermore, despite the presence of scientific literature on plastic-pollution-related policy measures in developing countries, policies targeting the contamination of agricultural land by microplastics are limited. For example, the review of [138] on plastic-related policies for Africa summarizes them, yet there are no policies that aim to restrict the contamination of agricultural land and plant intake.

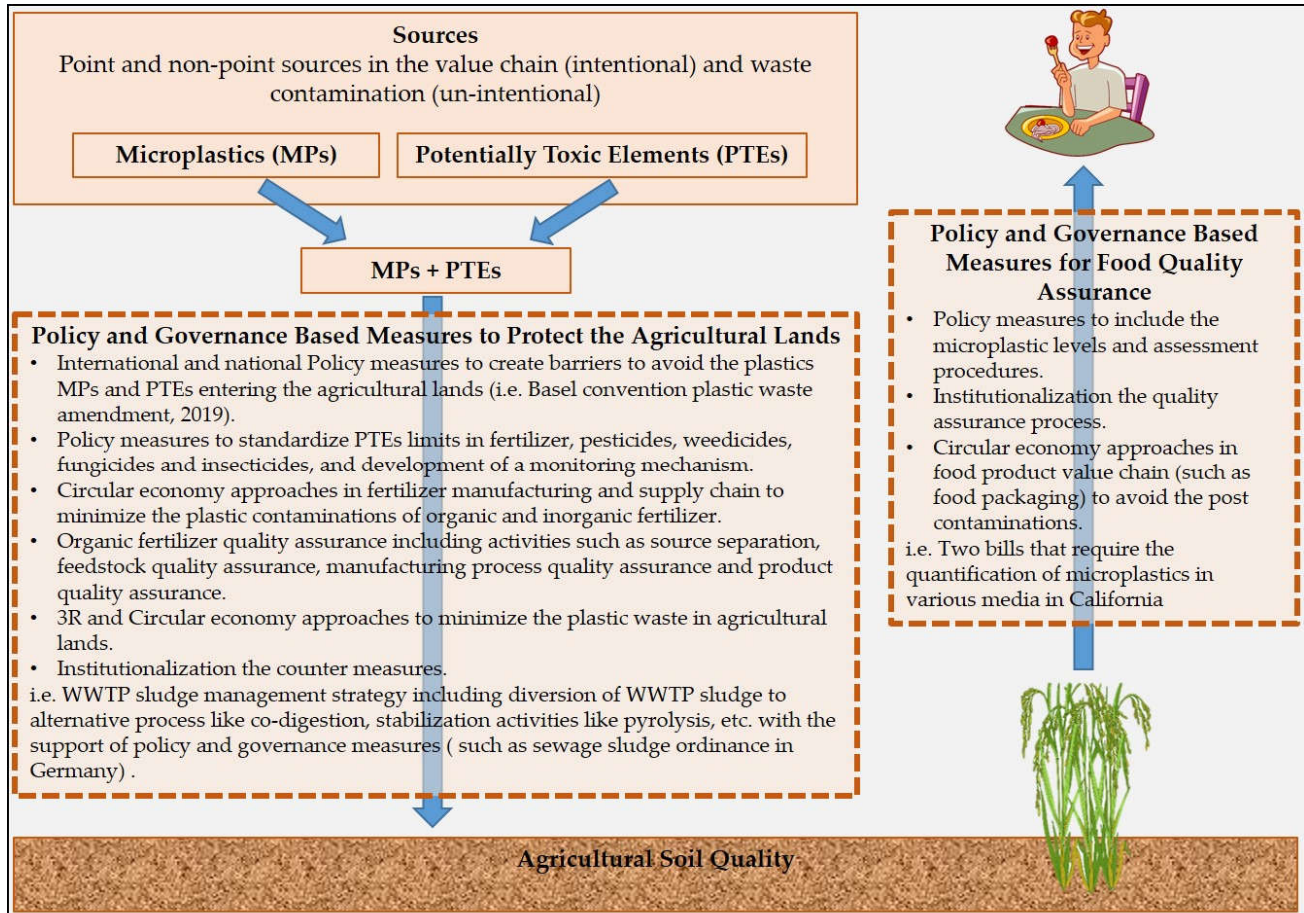


Figure 3. Policy and governance-based countermeasures.

Apart from sewage sludge, other forms of organic fertilizers, as discussed previously, can be a major source of microplastics due to contamination at the source, during production, or post-production, such as packaging. Since it is practically impossible to decontaminate microplastics during manufacturing processes, such as the composting of organic fertilizer, the feasible approach is minimization by implementing proper source separation. Furthermore, proper quality assurance measures should be taken to ensure feedstock quality, production process quality, and post-production quality.

Plastic-coated fertilizer (or controlled-released fertilizers, considering the function from agricultural perspectives) is a classic example of the direct intentional application of MPs into agricultural lands over the last two decades, the consequent effects and accumulations of which are now being reported [10]. In the context of environmental pollution, these plastic-coated fertilizers are reported to decrease total fertilizer usage by 20–30% and minimize the environmental leakage of nutrients while increasing productivity [139].

Hence, the implementation of policies and governance measures shall be considered the complete life cycle impacts of the products, process, and alternative scenarios. The measures shall target 3R- and various circular-economy-based approaches after assessing

the situation. Deng et al. [140] suggested policy measures to reduce the production of MPs and promote public willingness for recycling strategy applications. Apart from this, countermeasures must be for managing plastic litter (as a source of secondary MPs) and MPs with public participation, including citizen science contributions [141].

## 6. Conclusions

PTEs adsorb onto MPs through physicochemical interactions and increase their mobility in soils. Hence, MPs act as vectors for PTEs in soil. MP and PTE complexes can be easily adsorbed by plants. Therefore, the accumulation of PTEs in plants can be enhanced in both MP- and PTE-contaminated soils. Furthermore, there is a high risk of food chain contamination with PTEs by crops grown in both MP- and PTE-contaminated soils. Certain intentional (i.e., controlled-release fertilizer) and unintentional (i.e., Sewage sludge application), agricultural applications are identified sources of contamination of agricultural lands. Policy- and governance-based countermeasures shall focus on both agricultural land quality assurance and food quality assurance to minimize human exposure due to the ingestion of PTEs related to MPs.

**Author Contributions:** Conceptualization, A.D.I., M.G.Y.L.M., P.G., A.A., and P.J.D.G.; methodology, A.D.I.; investigation, A.D.I., M.G.Y.L.M., and P.G.; resources, A.A. and P.J.D.G.; data curation, A.D.I. and P.G.; writing—original draft preparation, A.D.I., M.G.Y.L.M., P.G., and A.A.; writing—review and editing, A.D.I., M.G.Y.L.M., P.G., A.A., P.J.D.G., M.T., M.O., T.F., and N.I.; visualization, A.D.I. and A.A.; supervision, A.A. and P.J.D.G.; project administration, A.A. Funding acquisition, P.J.D.G. and A.A. All authors have read and agreed to the published version of the manuscript.

**Funding:** The manuscript preparation expenses were partially funded by Institute for Global Environmental Strategies (IGES), Japan.

**Conflicts of Interest:** The authors declare no conflict of interest. The funders had no role in the design of the study; in the collection, analyses, or interpretation of data; in the writing of the manuscript, or in the decision to publish the review.

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